Pitfalls of Physical Habitat Simulation in the Instream Flow Incremental Methodology

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PITFALLS OF PHYSICAL HABITAT SIMULATION IN THE INSTREAM FLOW INCREMENTAL METHODOLOGY

by

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ABSTRACT

Shirvell, C. S. 1986. Pitfalls of physical habitat simulation in the instream flow incremental methodology. Can. Tech. Rep. Fish. Aquat. Sci. 1460: 68 p.

Physical habitat simulation (PHABSIM) in the Instream Flow Incremental Methodology may be ineffective for predicting the effects of streamflow regulation on fish populations. A review of eleven studies found that weighted usable area (WUA), the output from the PHABSIM model, was not related to fish abundance or biomass in most cases. This inconsistent relationship between fish biomass and WUA has two causes: 1) many implicit assumptions within PHABSIM are commonly not met in practice, and 2) weighted usable area, because it does not incorporate any aspect of an environment's productive capability, is an incomplete index of fish "habitat". This deficiency in WUA may be the main cause of the variable relationship because it is speculated that the productive capacity of an environment is more important in determining fish production than the absolute amount of WUA. The Instream Flow Incremental Methodology has many new options intended to improve its reliability but because these are not commonly in use yet, application of IFIM is mostly based on PHABSIM.

Key words: PHABSIM, IFIM, streamflow, regulation, fish habitat, microhabitat, hydraulic simulation

RESUME

Shirvell, C. S. 1986. Pitfalls of physical habitat simulation in the instream flow incremental methodology. Can. Tech. Rep. Fish. Aquat. Sci. 1460: 68 p.

La simulation de l'habitat physique (PHABSIM) utilisée en méthodologie par accroissement interne du débit (IFIM) peut s'avérer inefficace pour la prévision des effets de la régularisation de l'écoulement sur les populations de poisson. L'examen des résultats de onze études ont montré que la superficie poundérée utilisable (SPU), l'extrant du modèle PHABSIM, n'était généralement pas liée à l'abondance du poisson ou à la biomasse. Ce manque de cohérence entre la biomasse du poisson et la SPU a deux causes: 1) bon nombre d'hypothèses implicites du modèle ne s'appliquent pas en réalité et 2) la superficie pondérée utilisable, qui ne tient compte d'aucun aspect de la capacité productive de l'environnement, constitue un indice incomplet de "l'habitat" du poisson. Cette carence de la SPU pourrait bien être la principale cause de la variabilité de la relation, car on suppose que la capacité productive d'un environnement est plus importante que la valeur absolue de la SPU pour la détermination de la production piscicole. Il existe plusieurs nouvelles options permettant d'accroître la fiabilité de la méthodologie par accroissement interne du débit, mais comme celles-ci ne sont pas encore couramment utilisées, l'application de la IFIM repose surtout encoure sur le PHABSIM.

Mots clés: PHABSIM, IFIM, débit, régularisation, habitat du poisson, micro-habitat, simulation hydraulique

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INTRODUCTION

Conflict between offstream uses of water (such as hydroelectric generation where water is routed through a long penstock before being returned to the channel or agricultural irrigation) and instream uses (such as fisheries, navigation, or aesthetics) has rapidly increased as more projects which remove water from the stream channel have been built or proposed. To aid in the objective assessment of the consequences of water removal from streams, a current goal of fisheries research has been to quantify the relationship between fish production and streamflow (Fig. 1). As an initial step, methods have been developed which attempt to quantify the relationship between streamflow and fish habitat (relationship 2). There are now over twenty methods in use in Canada and the United States. Their use in Canada has been reviewed by Hatfield and Howard 1983, and Bietz and Campbell 1983, while their use in the United States has been reviewed by Loar and Sale 1981, and Wesche and Rechard 1980. These methods range from the simple to the complex, and based on their increasing data requirements, they lead to progressively costlier and more complex (not necessarily more accurate) analyses (Table 1).

The most popular and complex method now in existence is the Instream Flow Incremental Methodology (IFIM) developed over the last decade by the Instream Flow Group in Colorado (Bovee 1982). The Instream Flow Incremental Methodology is the most widely recognized method in use in the United States (Wesche and Rechard 1980).

It is accepted by some regulatory agencies as "the most scientifically and legally defensible [method] available for most instream flow problems" (U.S. Dept. of Interior 1979).

The main characteristics which account for its popularity and acceptance over other methods are that for the first time, it is a method which uses an objective, quantified definition of physical "habitat", and it has the ability to model the consequences to fish "habitat" of two or more different streamflows (changes of streamflow in increments). The method is based on the assumption that physical habitat would become limiting as flows were reduced and that changes in fish population abundance could be related to changes in specific physical components of the environment (Bovee and Cochnauer 1977). The cornerstone of the Instream Flow Incremental Methodology is a subset of computer programs called PHYSICAL HABITAT SIMULATION (PHABSIM) which calculates the amount of physical habitat available at different streamflows.

Despite the widespread use of PHABSIM, it has some critics who challenge the assumptions, logic, and mathematics of the method. This review is intended for those not familiar with PHABSIM application. It reviews how PHABSIM works, and assesses its limitations, successes, and failures. Specific objectives are:

1) to describe how PHABSIM calculates total fish habitat at various streamflows,

INSTREAM FLOW RELATIONSHIPS

PROBLEM: water removed for irrigation
industrial/domestic supply
hydroelectric generation

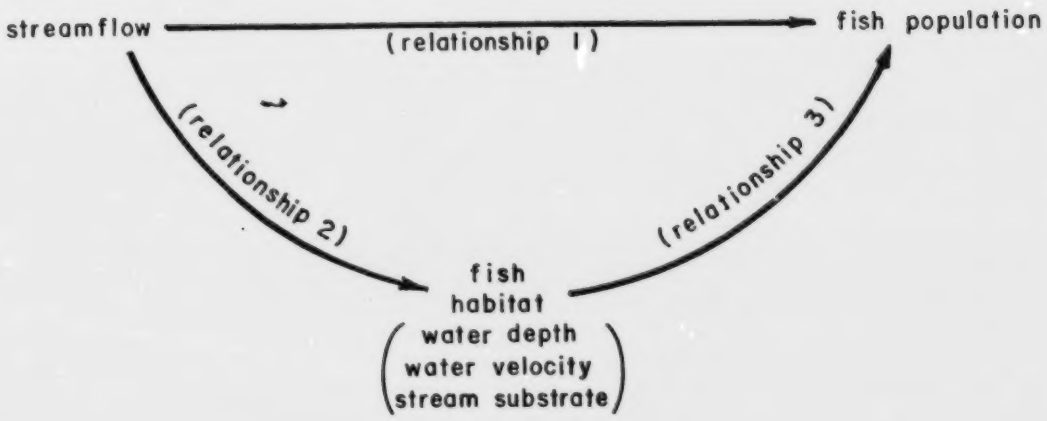


Fig. 1. Conceptual approach taken by fisheries researchers to determine the relationship between streamflow and fish populations.

Table 1. Instream flow assessment methods categorized by their data requirements.

Data requirements	Decision rule	Examples of method type	Reference	Data requirements	Cost	Level of resolution
Streamflow records	Percent of normal streamflow	Montana Method Constant Yield Method Flow Duration Method	Tennant 1976 U.S. Fish and Wildlife Service 1981 Anonymous 1974	low	low	low
Streamflow records + Transect data	Inflection point	U.S. Forest Service Transect Method WSP hydraulic simulation Usable Width Method	Bartschi 1976 Anonymous 1976 Sams and Pearson 1963			
Streamflow records + Multiple transect data + Fish habitat criteria	Weighted usable area	Instream Flow Incremental Methodology	Bovee 1982	high	high	high

2) to assess the validity of PHABSIM's assumptions,
3) to review tests of PHABSIM's accuracy, and
4) to discuss the current level of development of incremental-type methodologies.

HOW THE PHYSICAL HABITAT SIMULATION (PHABSIM) CALCULATES THE AMOUNT OF AVAILABLE FISH HABITAT

MICROHABITAT

The PHYSICAL HABITAT SIMULATION is a part of the Instream Flow Incremental Methodology designed to calculate quantitative changes in physical fish microhabitat due to increments of flow change. The underlying principles of PHABSIM are: each fish species exhibits preferences within the range of habitat conditions it can tolerate; these ranges can be defined for each species; and the area of stream providing these conditions can be quantified as a function of streamflow. Provided that the method is applicable for the river reach under investigation, PHABSIM calculates physical fish habitat close to the fish's position (called microhabitat) by three basic steps:

1) stream width, water velocity, water depth and substrate size within a reach of river are measured at known streamflow(s),

2) stream width, water velocity, and water depth of the reach are mathematically simulated for streamflows other than those measured, and

3) measured or assumed preferences for water depth, water velocity and substrate of the species or lifestage of interest are matched to the simulated widths, depths, and velocities available at the various streamflows.

The impact of reduced streamflow on the fish population is then inferred based on the calculated changes in the amount of suitable habitat available at the "new" streamflow.

MACROHABITAT

A necessary prerequisite before proceeding with a simulation of physical microhabitat conditions is to determine whether the macrohabitat of the stream reach is suitable for the fish species of interest. Sediment yield, chemical yield, channel structure, water temperature and water quality (collectively called macrohabitat variables) are considered in the typical Incremental Methodology application as shown in Figure 2. Models for predicting water quality at reduced streamflows have been proposed by Grenney

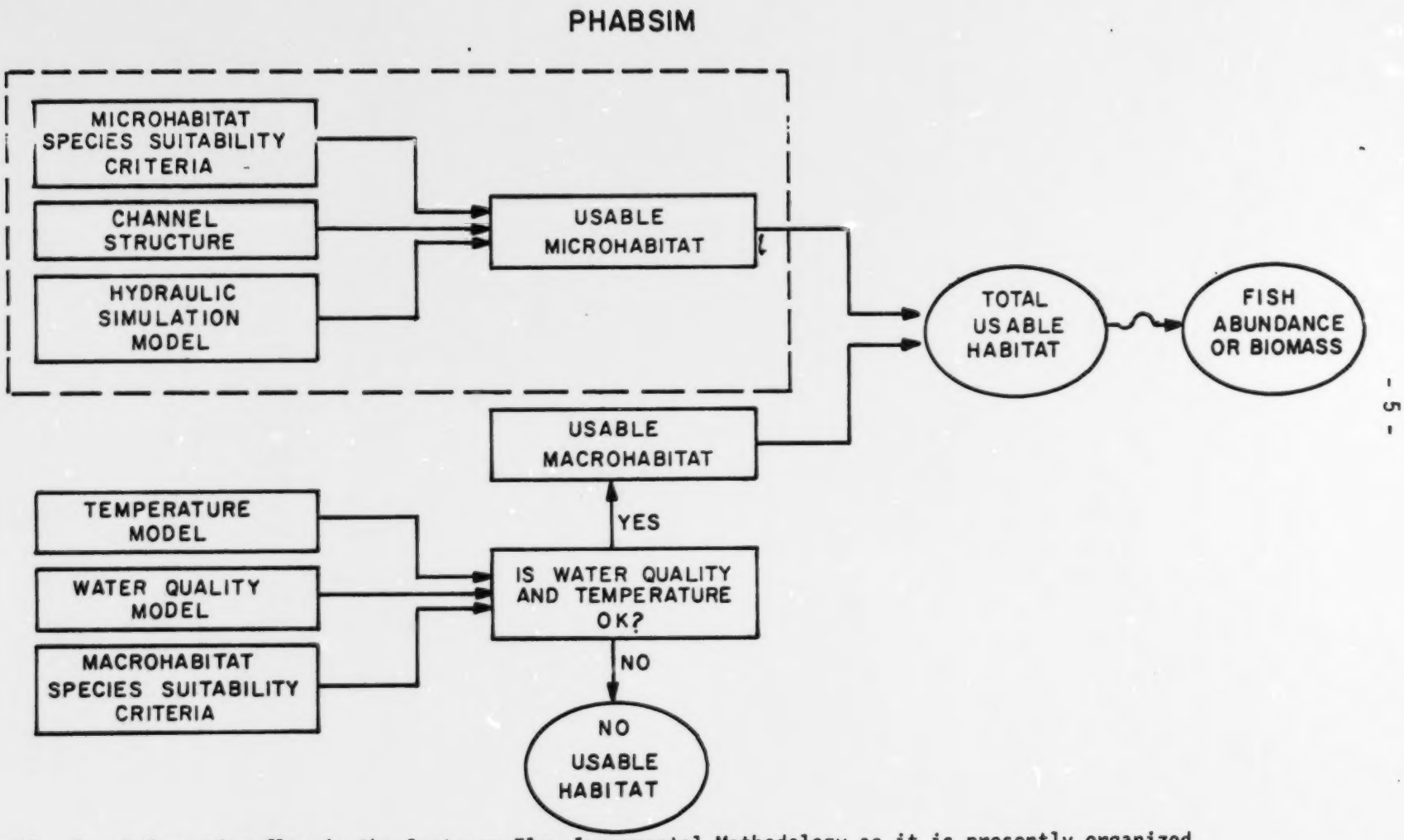


Fig. 2. Information flow in the Instream Flow Incremental Methodology as it is presently organized.

and Kraszewski (1981) and water temperatures models have been proposed by Theurer, Voos, and Miller (1984). These models have been developed to predict macrohabitat conditions at altered streamflows. Apparently these models are still theoretical at this time as no assessment of their predictive accuracy has been published. Results of Instream Flow Incremental Methodology applications published to date (Table 2) appear not to have used these parts of the method. Nevertheless, in theory, by superimposing the amount of usable microhabitat with usable macrohabitat, as estimate of total usable habitat is determined for a reach (Fig. 2). As a consequence, therefore, if the macrohabitat characteristics of the reach are currently unsuitable or are predicted to become unsuitable due to streamflow alteration, then PHYSICAL HABITAT SIMULATION is irrelevant for that reach because there will be no usable habitat.

MEASUREMENT OF HYDRAULIC CHARACTERISTICS AT KNOWN STREAMFLOW

Provided the stream reach is suitable for hydraulic simulation, (i.e. macrohabitat conditions will be suitable) hydraulic conditions are characterized at a known (calibration) streamflow from measurements taken along transects within the reach. The Instream Flow Incremental Methodology recognized two types of reaches for physical microhabitat simulation: "representative" reaches and "critical" reaches.

Representative reaches are similar to any other reach within an area - the hydraulic features of the reach are repetitive, and the reach length includes most of the hydraulic variance found in the entire section. Representative reaches are usually positioned by random selection and in theory, hydraulic simulations for representative reaches can be extrapolated to the entire river segment provided that at least 10% of the reach length has been measured (Bovee and Milhous 1978).

Critical reaches (sometimes referred to as unique reaches) are areas sensitive to changes in streamflow that contain localized, but rare habitat, essential for a particular life stage (for example, spawning habitat). Critical reaches are selected based on known fish concentrations, but because of their unique characteristics, hydraulic simulations for critical reaches cannot be extrapolated to the remainder of the river.

Transect placements within the reach is a critical factor determining the representativeness and reliability of hydraulic simulations (Bovee and Milhous 1978). Two types of transects are required for two different purposes:

1) hydraulic simulation transects, which are placed at and parallel to each and all of the reach's hydraulic controls (physical aspects of the streambed or banks which determine the height of the water surface upstream), and

2) habitat characterization transects, which are placed through the center of a habitat type so as to be representative of that habitat feature.

Table 2. Relationship between Weighted Usable Area calculated by PHABSIM and fish standing stocks.[1] Studies are ranked in descending order from good to poor relationships.

Author	Region	Species	Predictive Ability[2]
Anderson 1984	Colorado	brown trout	very good. $r^2 = 0.89$
Nelson et al. 1984	Pennsylvania	rock bass adults (Ambloplites rupestris)	positive, significant correlation, max $r^2 = 0.88$, very good.
Gowan 1984	Michigan	brown trout, 1984	very good. positive, significant correlation, $r^2 = 0.84$
Stalnaker 1979	Wyoming	brown trout (Salmo trutta)	very good. $r^2 = 0.81$
Orth & Maughan 1982	Oklahoma	stoneroller (Campostoma anomalum) freckled madtom (Noturus nocturnus) orangebelly darter (Etheostoma radiosum)	good, correlation but only in summer $r^2 = 0.47$ to 0.85
Loar et al. 1985	Tennessee North Carolina	brown trout	good 68 out of 160 (43%) significant, positive correlation, $r^2 = 0.41$
White et al. 1981	Oregon (artificial stream)	rainbow trout (Salmo gairdneri) summer, autumn low streamflow	average, positive relationship but nonlinear
Shirvell & Morantz 1983	Nova Scotia	Atlantic salmon, juvenile (Salmo salar)	poor, negative correlation, $r^2 = 0.1$ to 0.5
Orth & Maughan 1982	Oklahoma	smallmouth bass (Micropterus dolomieui) juvenile adult	poor, no significant correlation $r^2 = 0.0$ to 0.5
Loar et al. 1985	Tennessee North Carolina	rainbow trout	poor, 14 out of 160 (9%), significant, positive correlation, $r^2 = 0.45$
Loar et al. 1985	Tennessee North Carolina	brown trout	poor, 10 out of 160 (6%), significant, negative correlation, $r^2 = 0.40$
Irvine et al. (in press)	New Zealand (artificial streams)	rainbow trout, brown trout, chinook salmon (Oncorhynchus tshawytscha)	poor, no relationship
Nelson 1980b	Montana	brown trout	poor, Incremental Method predicted 50% optimum streamflow
Nelson et al. 1984	Pennsylvania	rock bass juveniles (Ambloplites rupestris)	poor, no significant correlation
White et al. 1981	Oregon (artificial stream)	rainbow trout, autumn, high streamflow	poor, negative correlation
Rimmer 1985	New Zealand (artificial streams)	rainbow trout, juveniles	poor, no relationship

Table 2 (cont'd)

Gowan 1984	Michigan	brown trout, 1983	poor, no relationship
Anderson 1984	Colorado	rainbow trout	poor, no relationship

[1]Different authors have used different units for fish standing crop: Anderson 1984 = kg·1000 ft^{-1}, Nelson et al. 1984 = kg·305 m^{-1}, Gowan 1984 = no. of trout·950 ft^{-1}, Stalnaker 1979 = kg·km^{-1}, Orth and Maughan 1982 = kg·ha^{-1}, Loar et al. 1985 = g·km^{-1}, White et al. 1981 = % of fish remaining·62.3 m^{-1}, Shirvell and Morantz 1983 = no. of salmon·m^{-2}, Irvine et al. = g·m^{-2}, Nelson 1980b = lb·mi^{-1}, Rimmer 1985 = g·100 m^{-2}.

[2]r^2 = coefficient of determination, the amount of variation in the standing crop due to variation in WUA.

Microhabitat variables - water depth, water velocity, and substrate size - are measured at intervals along the transect. These measured microhabitat variables are assumed to remain unchanged, part way up and downstream to the next transect. For the convenience of surveying and calculation, the transects are usually positioned so that this distance is halfway to the next transect (see Milhous et al. 1984). The length of that distance, times the width between measurements along the transect produces an area or cell (Fig. 3) in which hydraulic conditions are assumed to be equal to the conditions measured on the transect. These measurements and calculations are repeated for the remainder of the transects within the reach. A matrix of water depth, water velocity and substrate size distribution through the reach at the calibration streamflow is produced.

HYDRAULIC SIMULATION AT PREDICTED STREAMFLOW

When streamflow in the reach changes, the surface width, mean water depth, and mean water velocity of the stream reach changes in a predictable way provided the channel structure remains unaltered (Fig. 4). For streamflow different from the one measured, the Physical Habitat Simulation simulates the stream width, the water depth and the water velocity in each cell of the reach using either of two methods:

1) A least squares regression equation of the logarithm of water depth or water velocity versus the logarithm of streamflow (computer program IFG4) or;

2) Water Surface Profile (WSP) calculations using the Manning equation (computer program (IFG2).

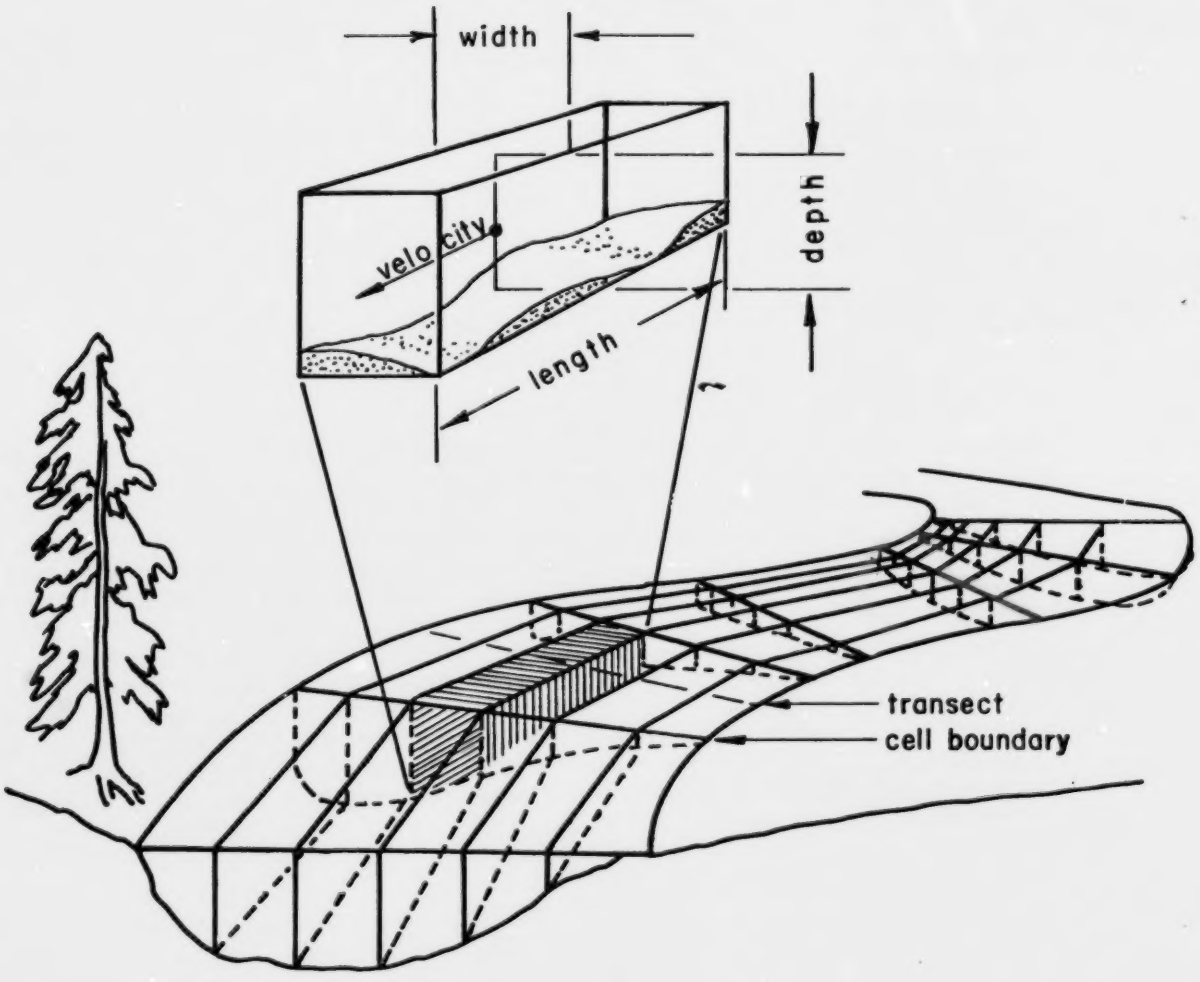
In addition, there are some optional hydraulic simulation techniques combining both WSP and regression analysis where WSP is used to simulate water depths and empirical velocity - streamflow relationships are used to simulate water velocities. These options are presented by Milhous et al. (1984), but no documentation for these computer programs currently exists and they are apparently rarely used in PHABSIM. The regression method (IFG4) is more accurate than the water surface profile method (Bovee and Milhous 1978; Wesche and Rechard 1980, p. 15) because it uses data collected at several streamflows. However, it also requires considerably more effort for data collection than the water surface profile which relies on data collected during one streamflow. Because applications of the Instream Flow Incremental Methodology normally use the Water Surface Profile calculation for its simulations, further discussion of PHABSIM hydraulic simulation methods are restricted to the Manning equation.

Manning's equation is:

$$Q = n^{-1} R^{2/3} S^{1/2} A, \qquad (1)$$

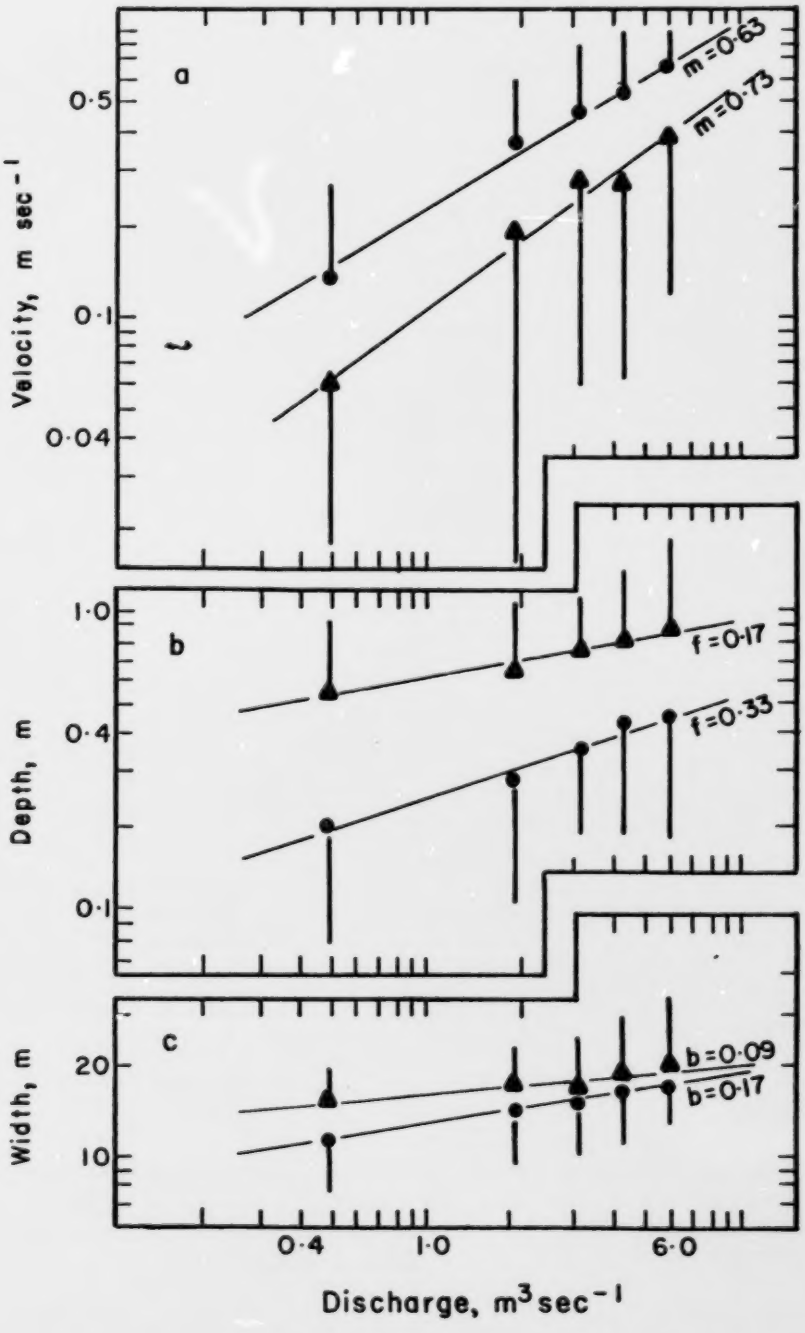
where Q = streamflow ($m^3 \cdot s^{-1}$),
R = hydraulic radius (m), or cross-sectional area divided by the wetted perimeter of the stream (roughly equivalent to the mean depth),

Fig. 3. Computer conceptualization of a stream reach showing transect position, assumed boundaries between hydraulic conditions measured at the transect, and hydraulic cells. Hydraulic parameters of water depth, water velocity, and substrate are measured at each transect subdivision and are assigned to the area of each cell.



$$\text{STREAMFLOW } (Q) = \sum (w \times d \times v)$$

Fig. 4. Comparison of rates of decrease of mean width, mean depth, and mean velocity of riffles and pools with decreasing streamflow for Deep Stream, New Zealand. The vertical bar is one standard deviation of the mean. Riffles ●; Pools ▲ (from Shirvell 1979); b, f, and m are the slopes of the regression lines specific to each variable (Rhodes 1977).



S = energy gradient, assumed parallel to slope of the water surface,
A = cross-sectional area (m^2), and
n = roughness coefficient, known as Manning's n, which may be calculated from stream measurements, or estimated from a description of bed materials, channel uniformity, and channel slope.

Since Q = VA (where V = mean water velocity in $m^3 \cdot s^{-1}$), Manning's equation can be transformed to

$$V = n^{-1} R^{2/3} S^{1/2} \qquad (2)$$

Manning's n can be solved for the calibration from

$$n = \frac{R^{2/3} S^{1/2}}{V} \qquad (3)$$

and is assumed to remain constant for hydraulic simulation at other streamflows. An option which allows n to be varied as a function of depth has been proposed by Milhous et al. (1984) but apparently it has not been widely used. Substituting streamflows (Q) for which hydraulic conditions have not been measured in equation 1 and solving for S, then substituting S, the new water surface into equation 2 and solving for V, the depth and velocity of each cell at different streamflows can be predicted. The theory of Manning's equation and examples of hydraulic simulation using this method are given in Bovee and Milhous (1978).

HABITAT PREFERENCES OF FISH

Once the hydraulic conditions in the reach have been simulated, the relative amounts of different hydraulic conditions available can be compared with the hydraulic preferences of the species or lifestage of interest. These hydraulic preferences are thought of as suitability criteria and are expressed as habitat preference curves. Habitat preference curves are based on three assumptions (Bovee and Cochnauer 1977):

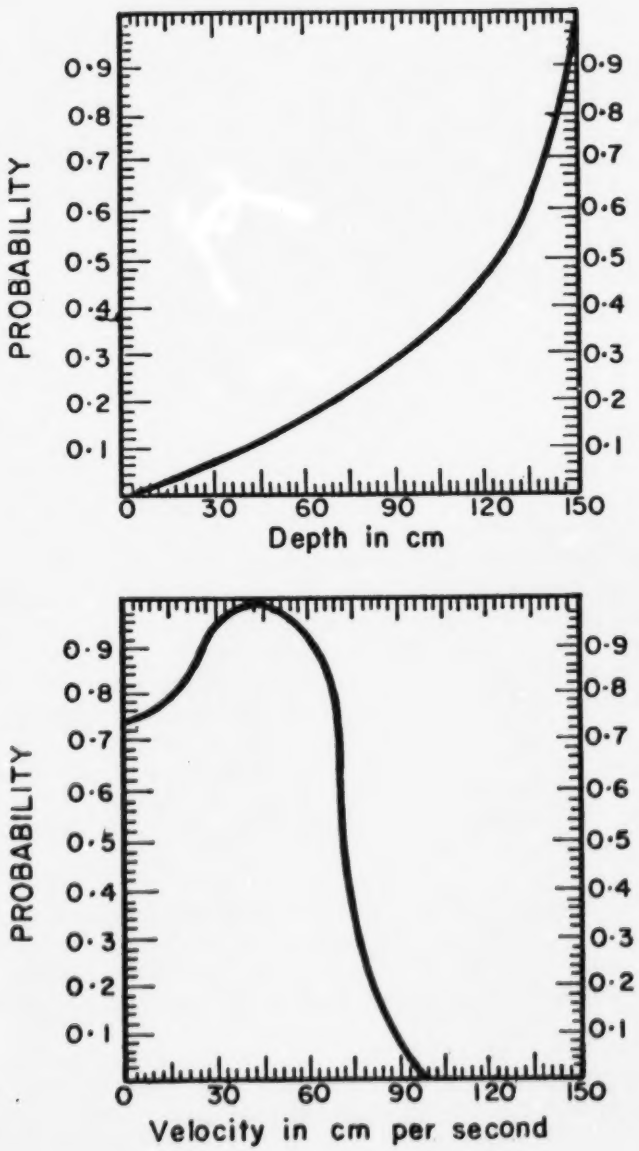
1) Individuals of a species will tend to select areas within the stream having the most favourable combinations of hydraulic conditions.

2) Individuals will also utilize less favourable conditions, with the probability of use decreasing with diminishing favorability of one or several hydraulic conditions.

3) Individuals will elect to leave an area before conditions become lethal.

Habitat suitability curves for variables of water depth, water velocity, and substrate are developed from frequency of occurrence histograms with the mode normalized to 1. An example of habitat suitability curves for adult smallmouth bass is given in Figure 5 (from Bovee 1978). There is

Fig. 5. "Probability-of-use" curves for adult smallmouth bass (from Bovee 1978).



considerable confusion over different names these curves have been given (see for example Mathur et al. 1985) based on the way they have been developed and their resulting implied interpretation (for a discussion see Armour, Fischer, and Terrell (1984). For Figure 5, many measurements of the water velocity and water depth occupied by adult smallmouth bass were taken. Bass were found to occur most frequently in water equal to, or deeper than 150 cm, and in water velocities of 43 $cm \cdot s^{-1}$. These levels of maximum occurrence are inferred to be the depths and velocities preferred by adults of this species (leading to the name preference curves, sometimes used), and the probability of using habitat with these preferred levels is assumed to be 1 (Bovee 1978). The Instream Flow Group has acknowledged that these are not true probabilities (pers. comm., Ken Bovee, IFG File 700.1) and has adopted the term suitability index curves.

For depths or velocities other than the preferred levels, the suitability is read directly from the curve. For example, the suitability for the depth of 105 cm is 0.37 (Fig. 5). The suitability for the velocity of 15 $cm \cdot s^{-1}$ is 0.81. The composite suitability for a depth of 105 cm and a velocity of 15 $cm \cdot s^{-1}$ is 0.37 x 0.81 = 0.30. Substrate size can be incorporated into this calculation from a suitability curve for substrate. In the preceding example, if the substrate found with that combination of depth and velocity had a suitability of 0.90, then the composite suitability of use for that combination of depth, velocity and substrate would be 0.37 x 0.81 x 0.90 = 0.27.

TOTAL USABLE HABITAT

Habitat preference curves incorporate the concept that certain levels of habitat variables are more usable than other levels by weighting the usable levels with proportionately higher index levels. Usable area is calculated by multiplying the weighted index value for the hydraulic conditions in each cell times the surface area of each cell to calculate the weighted usable area of the cell. This procedure equates a large area of marginal habitat (low preference) with a smaller, equivalent area of optimal habitat (high preference). For example, 100 m^2 of habitat with the depth, velocity, and substrate of the previous example would be equivalent to 100 (cell area) x 0.27 (composite suitability) = 27 m^2 of optimum habitat. The total usable area in a reach is calculated by summing the weighted usable area of each cell in the reach.

A worked example of an entire two dimensional matrix (depth and velocity) is given in Table 3. In each cell of the matrix, the upper number refers to the surface area of stream having a certain depth-velocity combination. The numbers in parentheses refer to the weighted usable area.

The total physical microhabitat in the reach at this streamflow is the sum of the weighted usable area of each depth-velocity combination, expressed as square meters of optimum habitat per kilometer of stream (lower right hand corner of the matrix). Due to the large number of calculations required for each matrix, the Instream Flow Incremental Methodology uses a computer program called PHABSIM to make the computations (Milhous, Wegner, and Waddle 1984).

Table 3. Distribution of depth-velocity combinations, expressed as m^2 per km of stream, and (in parantheses) weighted usable area (m^2/km) for adult smallmouth bass [from Bovee 1978].

Depth (m)	Velocity (m/s)								
	< 0.15	0.15-0.30	0.30-0.45	0.45-0.60	0.60-0.75	0.75-0.90	0.90-1.05	> 1.05	Totals
< 0.30	585 (22)	78 (4)							663 (26)
0.30-0.45	270 (24)	141 (15)		123 (14)	51 (5)	18 (1)	18 (0)	279 (0)	900 (58)
0.45-0.60	87 (11)	114 (17)	96 (15)	132 (21)	324 (38)	237 (5)	114 (0)	516 (0)	1,620 (107)
0.60-0.75	18 (3)	87 (17)	69 (15)	27 (6)	333 (53)	393 (11)	429 (0)	525 (0)	1,881 (105)
0.75-0.90	18 (4)	45 (11)	165 (44)	237 (63)	123 (24)	192 (7)	123 (0)	315 (0)	1,218 (153)
0.90-1.05	27 (7)	51 (15)	45 (15)	36 (9)	96 (23)	9 (<1)	447 (<1)		711 (69)
1.05-1.20	27 (9)	60 (23)		51 (21)	141 (43)	51 (3)	246 (3)		576 (102)
1.20-1.35		60 (29)		33 (17)	150 (58)	105 (7)	51 (1)		399 (112)
1.35-1.50		33 (22)		15 (11)	345 (189)	60 (6)			453 (228)
1.50-1.65				21 (20)	69 (50)	45 (6)			135 (76)
1.65-1.80		30 (27)			93 (68)	60 (8)			183 (103)
Totals	1,032 (80)	699 (180)	375 (89)	675 (182)	1,725 (551)	1,170 (53)	1,428 (4)	1,635 (0)	8,739 (1,139) *

* Total usable habitat per kilometer of stream.

RELATIONSHIP BETWEEN TOTAL USABLE HABITAT AND STREAMFLOW

A final step of the PHABSIM microhabitat model is to repeat this entire calculation of total usable habitat for a range of streamflows. Characteristically, the amount of usable habitat is minimal at low streamflows, increasing to a maximum near the median annual streamflow, then declining for further streamflow increases approaching bankfull (Fig. 6). In some cases, the amount of usable habitat increases continually with increasing streamflow (more water = more habitat) but generally the maximum usable habitat occurs somewhere near the median annual streamflow. Such a relationship is reasonable considering that fish species have evolved to maximize their fitness to habitat conditions (and streamflows) which occur most frequently.

Assumed changes in fish populations at reduced streamflows are inferred from the relationship between total usable habitat and streamflow. "...if there are no significant changes in nonphysical parameters, there will be a one-to-one ratio between the weighted usable area and standing crop" (Bovee 1978, p. 345). For example, using the habitat availability curve at the stream bottom in Figure 6, a reduction in streamflow from 1 $m^3 \cdot s^{-1}$ to 0.25 $m^3 \cdot s^{-1}$ reduces the total usable habitat from 26 m^2 to 18 m^2. If the number of Atlantic salmon fry are linearly related to the amount of total usable habitat at a one-to-one ratio, then it is inferred that the population would be reduced 31% by this streamflow reduction of 75%.

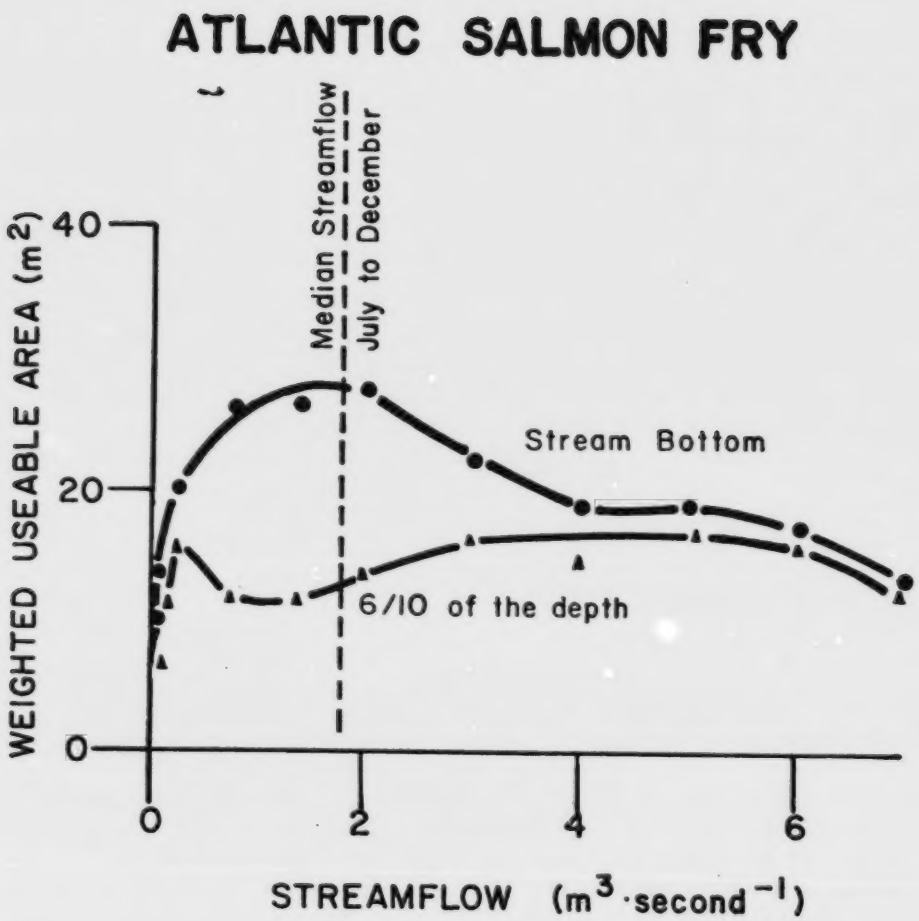
VALIDITY OF THE ASSUMPTIONS IN PHABSIM

To assess the reliability and effectiveness of the Instream Flow Incremental Methodology for assessing the effects of altered streamflows on fish populations, the validity of the basic assumptions underlying PHABSIM must be considered. Three general considerations are: (1) is PHABSIM appropriate for all rivers or fish populations? (2) having accepted that PHABSIM is applicable for the situation, are the mathematical procedures used to calculate total usable area valid? and (3) what is the relationship between weighted usable area and fish abundance or biomass?

REACH AND POPULATION SUITABILITY

A necessary prerequisite before proceeding with PHABSIM is to evaluate the specific reach and its fish population to determine whether the application is appropriate. Specifically, the macrohabitat conditions must be suitable and the fish population must be limited by the amount of physical microhabitat available. The Instream Flow Group identifies three conditions the reach must satisfy before PHABSIM is relevant (Bovee 1982, p. 4, 6):

Fig. 6. The amount of weighted usable area available to Atlantic salmon fry at the stream bottom (● - ●) compared with the amount of weighted usable area available at 6/10 of the stream depth (▲ - ▲) for streamflows less than bankfull (adapted from Shirvell and Morantz 1983).



1) The watershed must be in equilibrium (i.e. total sediment yield and chemical yield are in equilibrium with the total water yield from the watershed and the project will not affect this balance.

2) The channel structure of the reach will not be affected by the streamflow alteration.

3) Both the water temperature and water quality in the reach will be suitable for the fish species of interest following the streamflow alteration.

Increases or decreases in sediment will result in erosion or siltation in the reach. Changes in the chemical yield could cause changes in the water quality of the river, while decreases in streamflow frequently result in increased, and possibly lethal, water temperatures. Streamflow regime (the timing and magnitude of streamflow variation) is also important, as these are significant cues to fish for initiating certain activities (e.g. spawning migration, smolt emigration). Channel structure determines the relationships between streamflow and depth-velocity distributions within the reach.

Unfortunately, past experience has shown that streamflow regulation normally causes changes in channel morphology (Blench 1972; Buma and Day 1977; Simons 1979; Kellerhals 1982), water quality (Grenney et al. 1976; Shelton et al. 1978; MacCrimmon and Kelso 1970; Casey and Newton 1972, 1973; Shirvell 1979; Roux 1984; Wright 1984), and water temperature (Fraser 1972; Shirvell 1979; Ward and Stanford 1979; Gregoire and Champeau 1984; Wilson et al. in press). If these macrohabitat conditions are unsuitable for the species or are predicted to become unsuitable as a consequence of streamflow regulation, then it is pointless in proceeding with an analysis of the microhabitat conditions within the reach. It is therefore imperative that for an application of PHABSIM to be valid, the population must be limited by the amount of suitable microhabitat available. Secondly, if the channel structure is likely to change as a consequence of regulation, PHABSIM is again an improper assessment method because it is impossible to accurately simulate hydraulic conditions in the reach when you do not know what its final form will be.

ASSUMPTION OF IMPORTANT HABITAT VARIABLES

Once a user of the Instream Flow Incremental Methodology has determined that PHABSIM is valid for the reach, additional concern exists over whether PHABSIM is relevant for the fish populations of concern.

The suitability of application to specific populations depends upon the implicit assumption that water depth, water velocity, and substrate are the only, or at least the most important microhabitat variables determining habitat preferences of fish. This assumption is implicit in the method because at the present time these are the only variables which can be used in the calculations of weighted usable area. Also implicit in this assumption is that physical microhabitat and not food, water quality, competition, or predation are limiting the fish population, an assumption known to be false

for many fish populations (see Irvine et al. in press). Theoretically other variables should be able to be entered into the model provided they meet two criteria: (1) that they can be objectively quantified, and (2) that the way their influence changes with incremental changes in streamflow can be accurately modelled.

Some habitat variables known to be important in fish ecology may never be successfully incorporated into the model because they cannot successfully meet these two conditions. Two examples are cover and stream substrate. In PHABSIM, it is implicit that substrate or cover do not change with changes in streamflow. In the case of stream substrate, because there is no accurate method for predicting changes in substrate size due to changes in streamflow, the method assumes substrate composition remains constant. Although it is true that substrate size is relatively stable compared to water depth or water velocity for changes in streamflow, in the long term substrates do change (Kellerhals 1982) and can eventually have a profound effect on habitat suitability.

Likewise, while the concept of cover is well accepted as an important habitat element, the components of cover and how they change with changes in streamflow is poorly understood. Consequently cover is difficult to model accurately, and in the Incremental Methodology cover is assumed not to change with changes in streamflow. This is false, for example, when surface turbulence or water depth act as cover elements (see DeVore and White (1978) for turbulence and Stewart (1970) for depth), both of which change with streamflow variation (see Reynolds (1883) and Vogel (1981, p. 38-39) for changes in turbulence and Rhodes (1977) for changes in depth). The assumption that substrate and cover do not change with streamflow are clear departures from reality (see Wesche (1976) Fig. 1 for changes in cover), however, the magnitude of error introduced by this artificiality is unknown.

In any case, misidentification or inadequate inclusion of variables important for determining habitat suitability to fish could result in calculating stream conditions which are partly irrelevant to fish production. The consequence of not including factors such as food availability, predation, etc. is that the population must be limited by available microhabitat only (e.g. areas of suitable depths, velocities, and substrates), if the population is expected to respond to, and accurate predictions of change in fish populations are to be made from, changes in these variables alone.

ASSUMPTIONS IN HYDRAULIC SIMULATIONS

A fundamental assumption limiting the usefulness of hydraulic simulations is that hydraulic measurements and simulated responses to changes in streamflow at one location are representative of conditions and responses at another, and usually larger, area. The validity of this assumption depends partly upon the location where the calibration measurements are taken (i.e. the placement of the transect). For accurate hydraulic simulations, the transects must be placed at, and parallel to, all hydraulic controls within the reach. A hydraulic control is defined as a physical channel feature which

determines the height of the water surfaces upstream from that point at any streamflow (e.g. a riffle). Also, for the sample to be representative, the area measured should be "at least 10% of the total length of the stream in that segment" (Bovee 1982).

Four common conditions of transect placement frequently reduce the confidence one can place in the representativeness of hydraulic simulations:

i) Transects are either not placed at, or parallel to all hydraulic controls. Even when they are, some hydraulic controls (riffles) migrate upstream with decreasing streamflow (Bovee and Milhous 1978, p. 52).

ii) Some hydraulic controls are either missed on the initial survey, or their influence varies with changes in streamflow. At high streamflows, some hydraulic controls can be drowned out or their influence on the direction of streamflow can be reduced (Bovee and Milhous 1978, p. 56).

iii) Transects are placed within biologically critical but hydraulically unrepresentative reaches. (Critical reaches are defined as areas sensitive to changes in streamflow that contain localized, but scarce habitat, essential for a particular lifestage.) Because of their unique habitat characteristics, hydraulic simulations for critical reaches cannot be extrapolated to the remainder of the river.

iv) Less than 10% of the total stream reach is measured.

All of these common failures in transect placement can reduce both the accuracy of the simulations within the measured stream reach and limit the representativeness of those simulations to other areas of the river.

A second limitation to the Incremental Methodology's calculations is the assumption that Manning's n, used in the hydraulic simulations, remains constant at different streamflows (Bovee and Milhous 1978, p. 13). Manning's n varies from place to place and with streamflow (Bovee and Milhous 1978; Horton and Cochnauer 1980), but because Manning's n is measured for only one streamflow, the true value of n is not known with certainty for any streamflow other than for the calibration streamflow. Because Manning's n varies with streamflow, hydraulic simulations can be in error (the difference between the simulation and the actual measurement) up to a maximum of 133% - 200% (Bovee and Milhous 1978) depending upon the extent of extrapolation from the calibration flow. The mean error for 10 different streams was 39% (SD = 15%) when no limits were placed on the extent of extrapolation. When extrapolation was restricted to 40% - 250% of the calibration streamflow, the mean error was 19% (SD = 4%).

A third problem associated with changes in streamflow is the assumption that hydraulic conditions within each cell change in the same manner as the average change in these conditions for the entire cross-river transect. The relationship

$$v = aQ^b \qquad (4)$$

where v = velocity ($m \cdot s^{-1}$)
Q = streamflow ($m^3 \cdot s^{-1}$)
and a and b = intercept and slope respectively of the log-log regression equation, is well accepted for the mean channel velocity, but the form

$$v_i = a_i Q^{b_i} \quad (5)$$

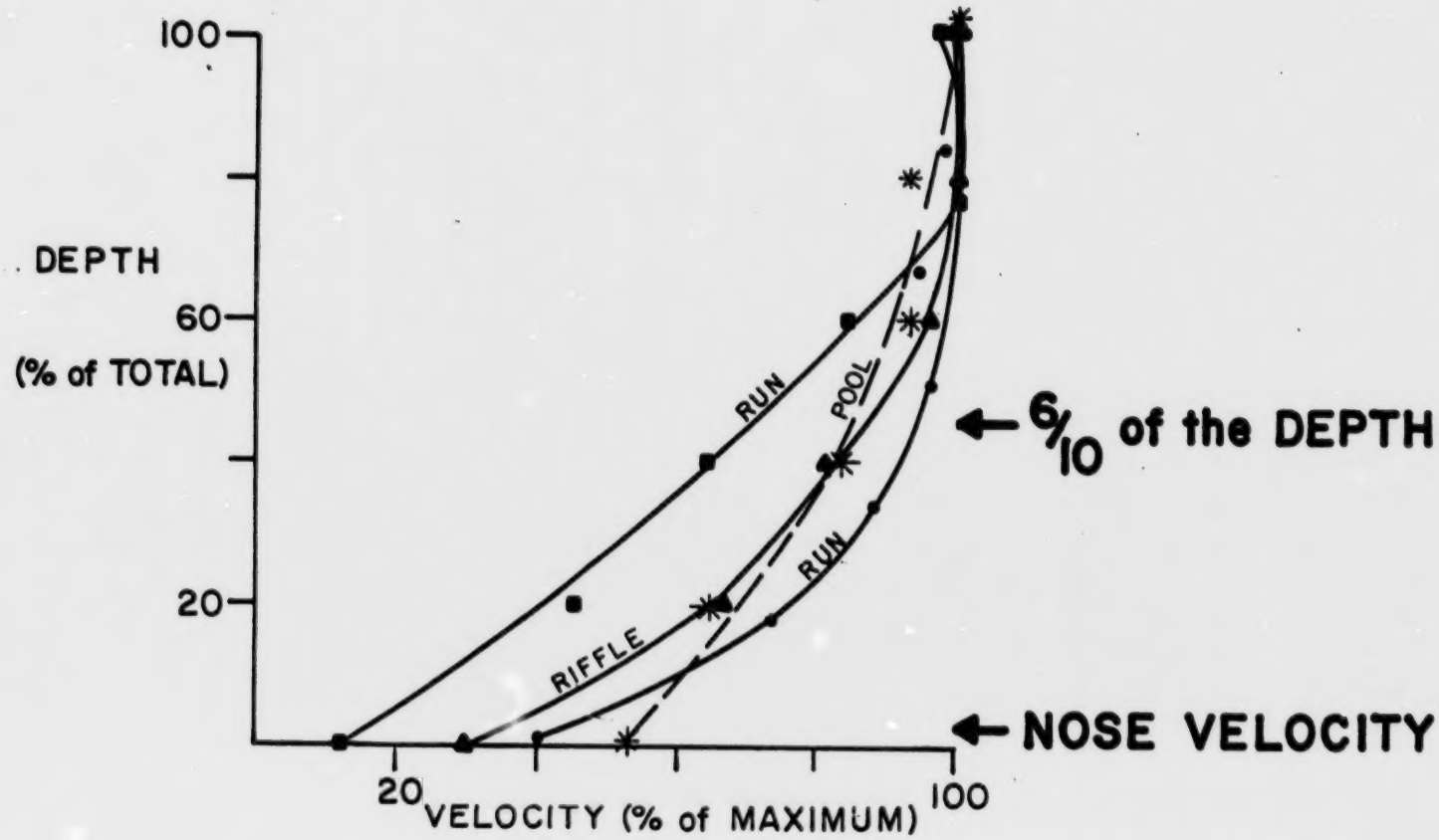
where all the symbols have the same meaning as in equation 4, except now for the i^{th} cell, is not generally accepted (Bovee and Milhous 1978). In an independent test of this assumption, Bovee et al. (1977) found that 64% of all simulations were in error by at least 10%, and 8% of all simulations were in error by at least 40%. The maximum error exceeded 60% and occurred in 4% of the simulations.

A fourth problem is the difference in water velocity between the position in the water column for which the hydraulic simulations are made, and the position actually occupied by the fish. Simulation of water velocity by PHABSIM IFG2 one flow measurement is always for the mean water column velocity (located at 6/10 of the depth from the surface). But most riverine fishes reside on or near the stream bottom. This results in PHABSIM simulating water velocities which in large rivers with mean depths greater than 1 meter can be from 50 cm to 100 cm above the positions the fish actually occupy. The difference in the amount of total usable habitat between these two locations can be 100% (Shirvell and Morantz 1983; Nelson 1980a). More importantly, when using 6/10-of-the-depth water velocities, maximum habitat occurs at lower streamflows (a mean of 47% less (Nelson 19890a)) than when using bottom velocities (i.e. it appears that more water can be removed from the channel before reductions in habitat occur).

The Instream Flow Group has attempted to eliminate this error by either simulating water velocities at the fish's position directly using equation 5, where the nose velocities are then subject to the same flaws as described for the hydraulic simulations, or by using various regression relationships between the water velocity at 6/10 of the depth and water velocity at the fish's position. However, estimating the velocity at the fish's position from the water velocity at 6/10 of the depth using a regression relationship introduces an additional error which is the difference between 1 and the coefficient of determination (r^2) of the regression relationship. Shirvell and Morantz (1983) found no consistent relationship between these two locations for positions occupied by juvenile Atlantic salmon. The mean decrease in water velocity at 6/10 of the depth to 2 cm above the stream bottom was 44% and varied depending upon the habitat type (run, riffle, or pool (Fig. 7)).

A fifth limitation of the hydraulic simulations is the extent which measurements of water depths and water velocities can be assumed to remain unchanged both across the stream, and up and downstream from the point of measurement. Instream Flow Incremental Methodology simulations assume that hydraulic conditions measured at a point on the cross-river transect remain unchanged both toward either bankside of that location and part way upstream and downstream to the next transect. Most users applying the method assume this distance to be half way to the next measurement along the transect, and half way upstream and downstream to the next transect. This means they assume

Fig. 7. The profile of water velocity with water depth for run, riffle, and pool habitat in the Beaver River, Nova Scotia.



that the depth and velocity measured at a particular location in the stream are constant throughout a rectangular cell (see Fig. 3). This assumption is rarely true, and the error it introduces becomes larger with the increasing size of the cells (i.e. the interval between measurements along the transect and the distance between transects). Researchers using PHABSIM usually do not report the size of the cells they used for hydraulic simulation, however, for small rivers, common cell sizes range from 26 m^2 (Trihey 1981 for pink salmon studies) down to 2.6 m^2 (Shirvell and Morantz 1983 for Atlantic salmon studies).

There are no standardized criteria for cell size with which to judge hydraulic simulations. The amount of departure from reality caused by large cell sizes depends upon how rapidly water velocities and depths change across and downstream - functions which may have slower rates of change in larger rivers. If this is so, then it may be possible to obtain acceptable accuracy for hydraulic simulations in larger rivers using larger cell sizes than in small rivers. The United States Geological Survey (Buchanan and Somers 1968) recommends that the mean cell width should not exceed 1/20 of the mean river width (i.e. there should be at least 20 measurements across the transect). Even though the sample size is small, the mean ratio of cell width to river width used by a survey of published incremental-type methodology studies was 20.5 (Fig. 8).

As an estimate of error[1] across a transect in a small Atlantic salmon river, the mean error at the cell edges was 15% for the velocity measurements and 14% for the depth measurements when the cell width was 1/30 of the river width. When cell width was 1/8 of the river width, the mean error at the cell edges increased to 29% for velocity and 36% for depth (Fig. 9).

Likewise, the error at the end of the cell downstream from the transect in a small Atlantic salmon river increased as the distance between the transects increased (Fig. 10). Riffles tend to occur every 5 to 7 stream widths (Hynes 1970, p. 15), therefore the maximum error of assuming the depth or velocity at the end of the cell is the same as at the middle of the cell should occur when cell lengths (i.e. the distance between transects) are 2.5 to 3.5 river widths apart. Transects placed further apart than 2.5 - 3.5 river widths will have less discrepancy between the depths or velocities at the end of the cells and the point of measurement because the depths are decreasing and the velocities are increasing as the next riffle is approached. The consequence of this cycling nature of depths and velocities downstream in a river is that the difference between depths or velocities at the end of the cell and the depth or velocities at the point of measurement is maximum at 2.5 - 3.5 river widths, and is asymptotic at distances greater than that.

As an estimate of error downstream from a transect in a small Atlantic salmon river, the mean error at the end of the cell was 41% for the depth measurements and 27% for the velocity measurements when the cell length was 0.8 river widths. When the cell length was increased to 2.7 river widths, the mean error at the end of the cell increased to 92% for depth and 47% for velocity (Fig. 10).

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[1]This is not the error of simulation; it is the error of assuming that the depth or velocity at the edge or end of the cell is the same as in the center of the cell, and is in addition to the error of simulation.

Fig. 8. Cell sizes used for hydraulic simulations in incremental-type methodology studies in small rivers. (Note: [b]horizontal and [c]vertical scales are not proportional.) [a]Trihey 1981; [b]Workman 1976; [c]Dooley 1976; [d]Orth (pers. comm. 1984); [e]Shirvell and Morantz 1983; [f]WR = ratio of mean river width to mean cell width.

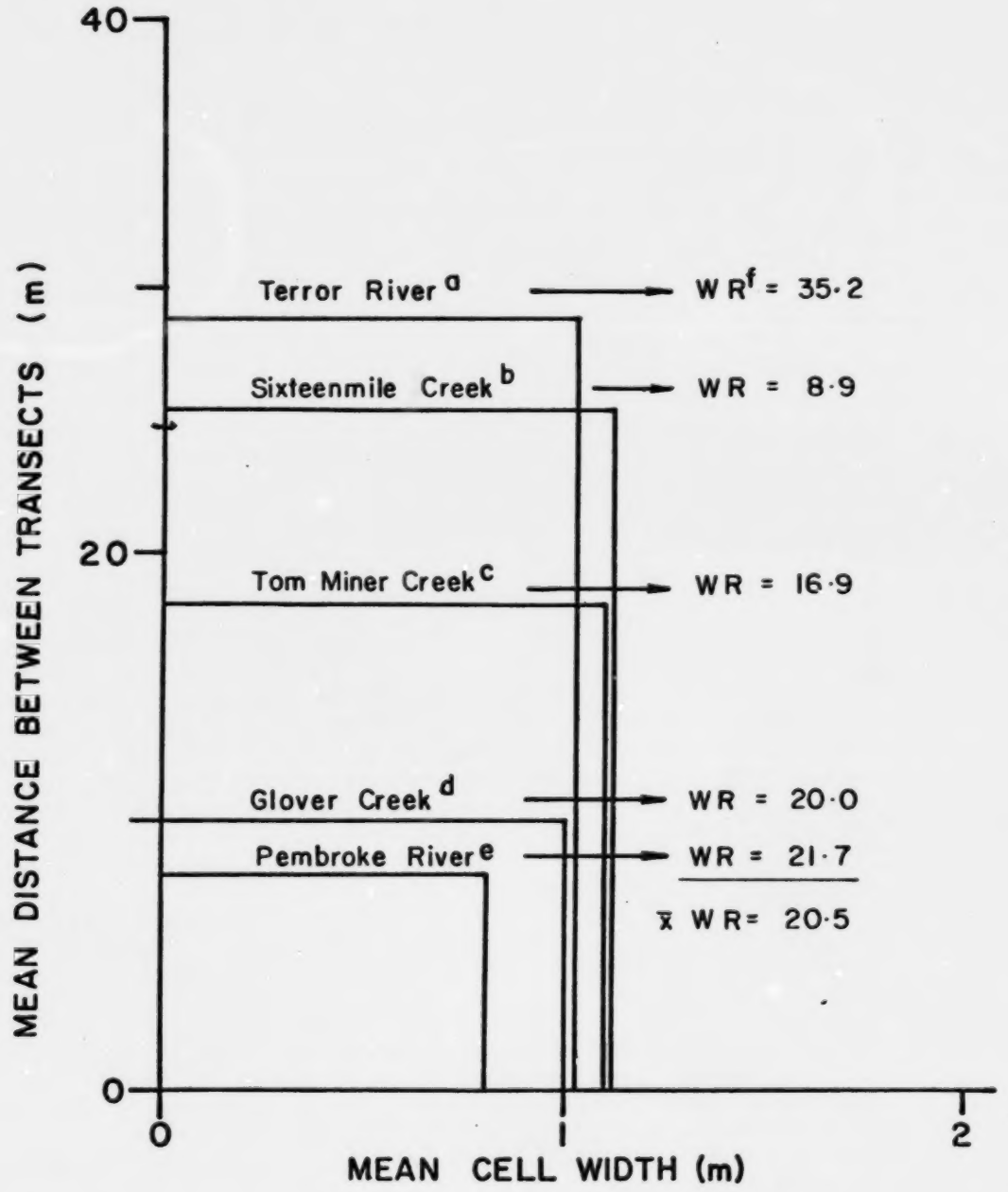


Fig. 9. Relationship between the mean error in the dpeth or velocity at the edge of the cell across the transect with increasing cell width (based on data from Tables 1, 2, 3, 4, 5, 6; Appendix 2; lines fitted by eye).

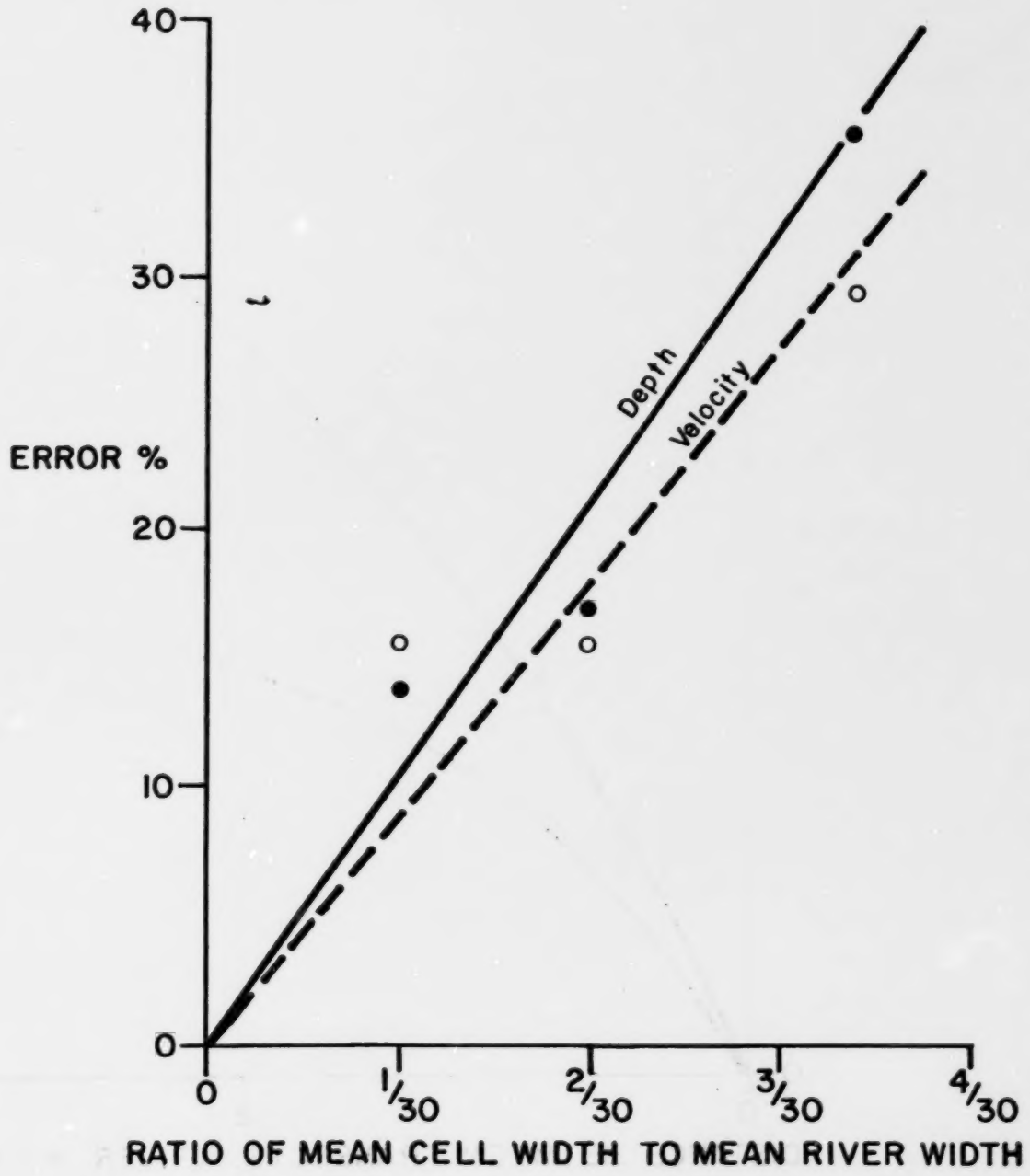
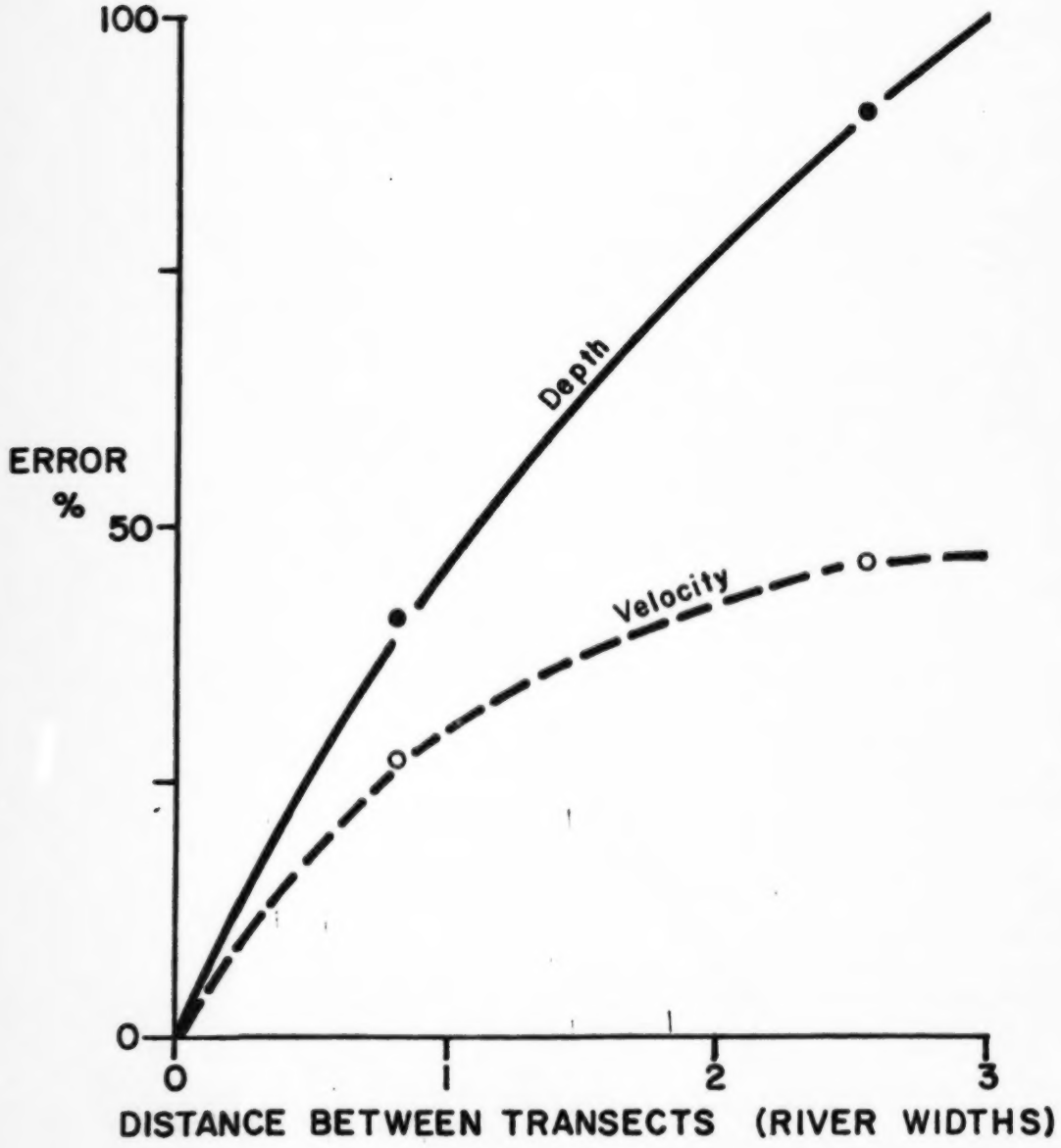


Fig. 10. Relationship between the mean error in the depth and velocity at the end of the cell downstream from the transect with increasing cell length (based on data from Tables 7, 8, 9, 10; Appendix 2; lines fitted by eye).



ASSUMPTIONS IN HABITAT PREFERENCE CURVES

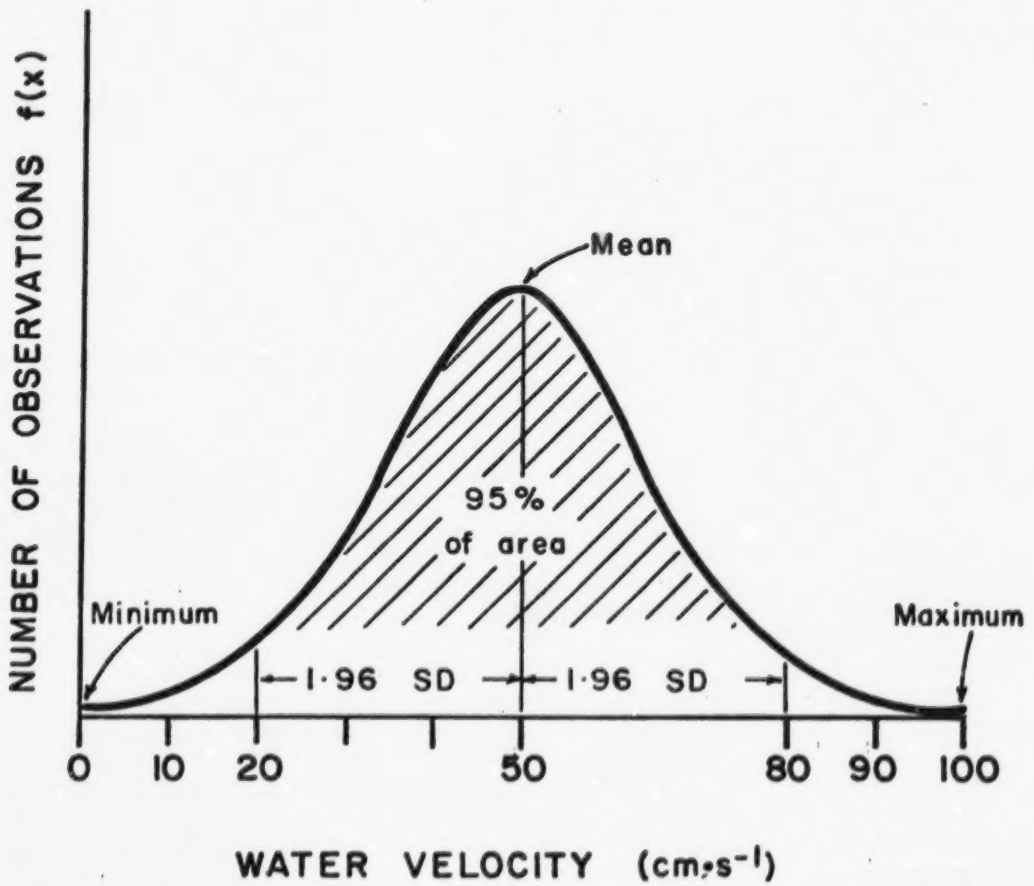
The first limitation to habitat preference curves is the incorrect interpretation of the curves as mathematical probabilities of fish occurrence. From normal statistics we know that the entire area under a frequency distribution curve is 1 and that 0.95 of the area (= 95% of the observations) is within ±1.96 standard deviations (SD) of the mean (Fig. 11). For Figure 11, this means there is a 99.999...% probability that fish will occupy positions with water velocities between 10 - 90 $cm \cdot s^{-1}$ and a 95% probability that fish will occupy positions with water velocities between 20 - 80 $cm \cdot s^{-1}$. The Instream Flow Incremental Methodology interprets the probability of fish occupying positions with the modal water velocities (exactly 50 $cm \cdot s^{-1}$) as 100%, but in fact, the probability of this low - most individuals are occupying water velocities other than 50 $cm \cdot s^{-1}$ but within the range of 10 - 90 $cm \cdot s^{-1}$.

This has important consequences in calculating total usable area. Referring back to the smallmouth bass example (Fig. 5), and using the Instream Flow Group interpretation of these curves, the probability of adult smallmouth bass occupying water 150 cm deep flowing at 43 $cm \cdot s^{-1}$ is 1 x 1 = 1 (i.e. absolutely certain). However, in reality, it is only absolutely certain (probability = 100%) that adult smallmouth bass will occupy water 0 - 150 cm deep flowing at 0 - 100 $cm \cdot s^{-1}$ (i.e. the sum of all distribution frequencies).

The Instream Flow Incremental Methodology has been critized for this interpretation before (Ken Bovee pers. comm., File: IFG 700.1) and the Instream Flow Group now refers to these curves as suitability index curves (Armour, Fischer, and Terrell 1984). Some users of the Instream Flow Incremental Methodology continue to refer to these curves as probability curves (e.g. White, Milligan, and Bingham 1981; Glova 1982; Anderson 1984; Sheppard and Johnson 1985) and all users, because they multiply the weighting factors of several habitat variables together continue to manipulate and interpret these curves as if they were probabilities (Bovee 1982, p. 176) regardless of what they call them.

A second, and associated, problem with the habitat preference curves is the implicit assumption that habitat variables are independent in their influence on habitat selection by fishes. Because the Instream Flow Incremental Methodoolgy multiplies the "probability" of using a certain value of one habitat variable, times the "probability" of using a certain value of another habitat variable, it implicitly assumes that fish perceive and judge the suitability of variables independently in their selection of habitat. This assumption has been found invalid for certain warm water fishes (Orth and Maughan 1982) and for brown trout (Shirvell and Dungey 1983). Mathur et al. (1985) observed that interaction between habitat variables can by itself explain 30% of the known variation in fish numbers and can be the factor most strongly related to fish abundance. The IFG suggests that variable interaction can cause error in the WUA estimate in the 6% to 15% range (IFG pers. comm. 1985) and Mathur et al. (1985) gives an example of how this interaction can lead to illogical conclusions about fish abundance when the assumption of independence is not met. Because this assumption of independence of microhabitat variables has been shown to be false for

Fig. 11. Hypothetical frequency distribution curve of water velocities occupied by a fish species.



certain species, it is possible this assumption may also be false for other species as well.

A third limitation of habitat preference curves is their inappropriateness for use with populations and environments other than those which they were derived. Bovee (1982) suggested that correctly developed preference curves should have universal applicability and Shirvell and Dungey (1983) found that brown trout from five dissimilar rivers had similar water velocity preferences. Dave Morantz (pers. comm. (Appendix 1, Table 1)) found that Atlantic salmon juveniles from five dissimilar rivers also had similar water velocity preferences but they had significantly different water depth preferences. Because different populations of the same species may have different preference curves for the same habitat variable other than water velocity, the Instream Flow Group now recommends that site-specific preference curves be developed before accepting their use as valid.

A fourth, and related limitation to the universality of habitat preference curves is the confusion between real versus apparent differences in preference. Habitat conditions preferred by a single population of fish can appear to be different depending on the method used to determine the fish's location (Horton and Cochnauer 1980) (e.g. explosives (Everest and Chapman 1972), electrofishing (Horton and Cochnauer 1989), gill nets (Bovee and Cochnauer 1977), underwater traps (Shepherd 1978), or direct underwater or surface observation (Gosse and Helm 1981; Shirvell and Dungey 1983, respectively)). These different sampling methods can result in <u>apparent</u> differences in habitat preference curves for the same population (Bain et al. 1982; Horton and Cochnauer 1980). At the same time, habitat conditions preferred by a single population of fish determined by the same technique can have real differences depending upon the season (Rimmer, Paim, and Saunders 1984), time of day (Helfman 1981), activity of the fish (Shirvell and Dungey 1983) or size of the individual (Everest and Chapman 1972; Shirvell and Dungey 1983). Apparent differences in habitat preference due to different techniques of fish capture may be confused with real differences attributable to different preferences for different activities (i.e. the sampling technique may cause the fish to change activities, for example from feeding to hiding, so that data thought to represent habitat preferences for feeding may really represent habitat preferences for hiding (see Horton and Cochnauer 1980, p. 26; Larimore and Garrels 1985)). The user of the Instream Flow Incremental Methodology must use astute judgment when selecting the correct preference curve to use or when sampling fish locations to produce his own curves because the output from PHABSIM is sensitive to the habitat preference criteria used. Most habitat preference curves used today are daytime habitat preferences for feeding during summer, yet these curves are frequently extrapolated to recommending streamflows for throughout the year.

ASSUMPTIONS IN THE CALCULATION OF WEIGHTED USABLE AREA

A unique characteristic of the habitat preference curves is the weighting of the fish's preference for certain level of habitat variables more than others. This is an implicit assessment that certain areas of the stream are more usable than others. This interpretation is based upon the assumption

that individuals "will utilize less favourable conditions, (but) with the probability-of-use decreasing with diminishing favorability..." (Bovee and Cochnauer 1977)[1]. Accepting that this assumption is true, PHABSIM calculates weighted usable area by multiplying the weighting factor times the area of stream with those conditions (e.g. a preference of 0.5 x 10 m^2 = 5 m^2 of usable habitat). The consequences of this calculation is that it implicitly assumes that 10 m^2 of partly (half) usable habitat will support the same biomass of fish as 5 m^2 of perfectly usable habitat. This assumption may fail if the relationship between the fish's perception of habitat desirability and the habitat's productive capability is nonlinear. For example, Baldes and Vincent (1969) found that brown trout would not use perfectly preferred habitat conditions once the area with those conditions became smaller than 0.14 m^2.

More importantly, this calculation of weighted usable area implies that areas of the steam with non-preferred conditions (e.g. weighting = 0; 0 x 10 m^2 = 0 m^2 of usable habitat) are useless. That is, that areas of the stream not directly occupied by the fish (or water that maintains unusable cells) can be eliminated with no effect on the fish population. Ecologists may reject this implicit assumption because non-occupied areas may produce food, provide channel or water quality maintenance, or contribute some other important, but poorly recognized function in the ecosystem.

A second limitation in the calculation of weighted usable area is that some users of the Instream Flow Incremental Methodology express usable habitat as a percent of the wetted area of stream bottom rather than as absolute amounts (e.g. m^2, hectares) of usable habitat (e.g. Orth and Maughan 1982). The use of percentages distorts the relationship of how weighted usable area changes with changes in streamflow, causing the maximum usable habitat to appear to occur at streamflows lower than it really does (Fig. 12). This is because as streamflow decreases, stream width decreases at a faster rate than the area of suitable habitat (because stream width decreases continually, but some habitat unsuitable at one streamflow can become suitable at a lower streamflow). Even though the absolute amount of suitable habitat has decreased, a greater proportion of what is left is suitable. The effect, then, of expressing total usable area as a percent of the wetted area is to cause the maximum usable habitat to appear to occur at a lower streamflow than it really does. In some cases, this difference in streamflow where maximum habitat occurs can be 300% (Fig. 12).

ASSUMPTION THAT THE POPULATION IS SPACE LIMITED

A final limitation of the Instream Flow Incremental Methodology lies in the variability of the relationship between weighted usable area calculated by PHABSIM and the number of fish in a stream. It is implicitly inferred that

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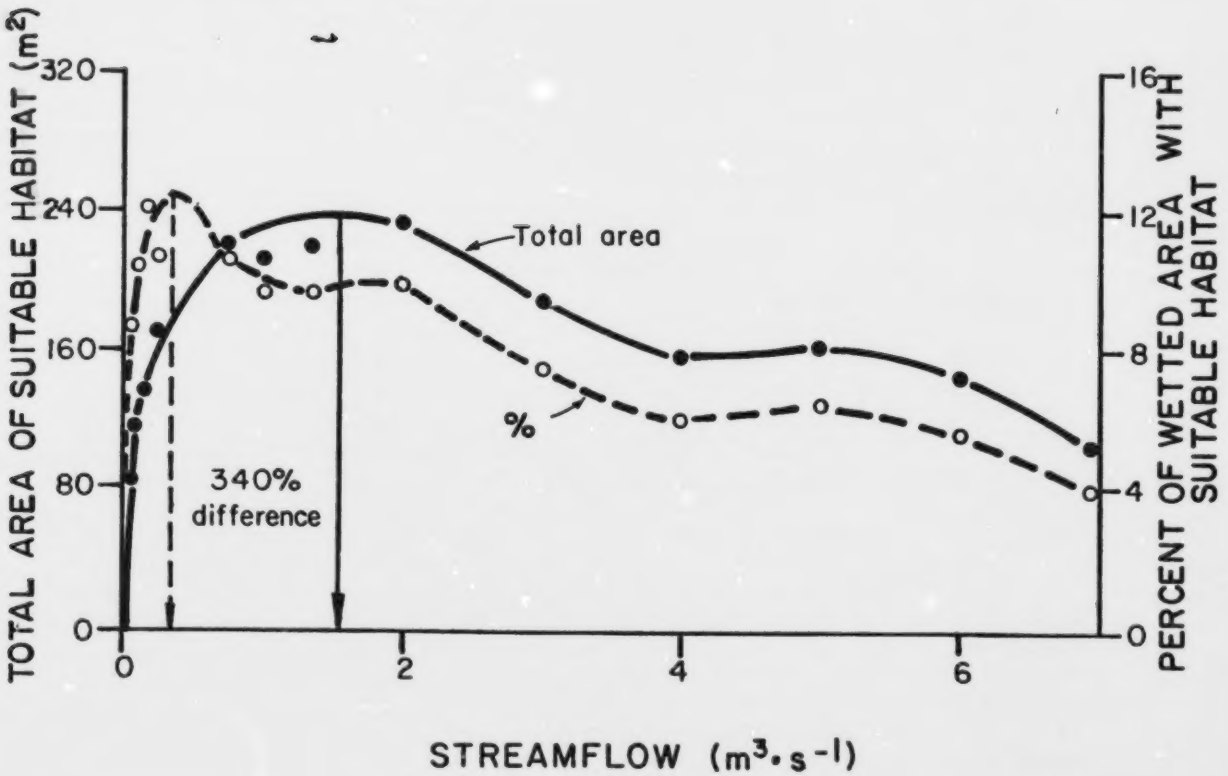
[1]While this assumption is generally accepted to be true, Bain et al. (1982) present evidence which they interpret to indicate that some age classes of fish remain in the same position regardless of changes in the desirability of conditions at that position.

Fig. 12. Comparison of suitable habitat expressed as total area with suitable habitat expressed as a percent of wetted area in the Pembroke River. Data are for Atlantic salmon fry habitat using stream bottom velocities. When suitable habitat is expressed as a percent of wetted area, maximum suitable habitat appears to occur at a streamflow 88% lower than the streamflow where absolute maximum suitable habitat (m^2) actually occurs (adapted from Shirvell and Morantz 1983).



there is a direct linear relationship between the amount of weighted usable area and a fish population's abundance or biomass. Users of the Instream Flow Incremental Methodology base their assessments of the impact of streamflow changes upon fish stocks on the change in the total amount of weighted usable area available at the "new" streamflow: increasing the amount of weighted usable area increases fish abundance, while decreasing the amount of weighted usable area decreases fish abundance (Bovee 1978, p. 345). This relationship is fundamental to the Instream Flow Incremental Methodology's usefulness because if this assumption is true, then fish stocks can be adequately protected or enhanced by preserving or creating weighted usable area through streamflow manipulations.

Several researchers have attempted to validate and quantify this relationship between the amount of weighted usable area and the abundance of fish stocks. While there is no doubt that a definite positive relationship exists between streamflow and the number of fish a stream can support (e.g. Kraft 1972; Nickelson and Reisenbichler 1977; Burns 1971; Allen 1969; Shirvell 1979), studies attempting to quantify the amount of weighted usable area with a population's size using the Instream Flow Incremental Methodology have had varied success (Table 2). Some studies (Stalnaker 1979; Orth and Maughan 1982) found good relationships as strong as $r^2 > 0.80$, while others found poor or even negative relationships (Nelson 1980b; Shirvell and Morantz 1983; $r^2 = 0.5$, $b = -1.5$). However, based on research comparing different methods for assessing reduced streamflow impacts, Nelson (1980b) concluded that "the IFG incremental method in its present state of development is not a consistent method for deriving instream flow recommendations for the trout rivers of Montana", while Shirvell and Morantz (1983) found that "the Incremental Methodology failed to predict accurately the amount of habitat...expected to have been available based on the numbers of Atlantic salmon (actually) present."

The best relationships between weighted usable area and fish populations have been for the biomass of populations in steady state with their environments (presumably at carrying capacity) before a perturbation of streamflow. The relationship between weighted usable area and fish numbers appears to be strongest for salmonids at life stages where space is the factor clearly controlling their abundance (e.g. area of spawning habitat (see Wickett 1958)). However, no relationship between fish biomass or abundance and weighted usable area has ever been demonstrated after a perturbation. And no relationship between fish production and weighted usable area has ever been demonstrated either before or after a streamflow perturbation.

Because of the mixed success of attempting to relate weighted usable area to number of fish, the significance of the analysis produced by PHABSIM continues to be controversial. The cumulative effect of the assumptions in PHABSIM is that even after having accepted that fish stocks are linearly related to the amount of fish habitat and that fish habitat is a function only of water velocity, depth, and substrate, the mathematical procedures of PHABSIM can potentially result in very large computational errors of the total weighted usable area available to fish at different streamflows.

LEVEL OF DEVELOPMENT OF THE INSTREAM FLOW INCREMENTAL METHODOLOGY

The Instream Flow Incremental Methodology has made good progress on quantifying the relationship between streamflow and water velocity and depth profiles through the stream (relationship 2; Fig. 1). The greatest advances have come from the addition of two important concepts to this relationship: first is the concept of "usability". The recognition that part of the stream is usable implies that the remaining part is unusable - a simple, but important understanding of fish habitat when comparing the productivity of different streams or the same stream at different streamflows. The second important concept is that some of the usable habitat is more usable than others (the weighting of usability). This concept recognizes that a small gain or loss of critical, rare habitat could have the same consequences to the fish population as a large gain or loss of unusable stream area. This concept recognizes that a population's production may depend largely on a small amount of critical habitat, and that even a small reduction in usable habitat could potentially have an unacceptable effect on the entire population.

POOR RELATIONSHIP BETWEEN FISH BIOMASS AND WUA

However, while the IFIM has had relatively good success quantifying the effects of streamflow changes on river hydraulics, no progress has been made on the relationship between weighted usable area and the fish population (relationship 3; Fig. 1). Although this is fundamental to the use of IFIM, it is apparently accepted on blind faith that there is a direct linear relationship between weighted usable area and fish abundance. "In no case in the literature or from the interviews could the consultants find situations where the flow regime decided upon, as a result of using any of the methodologies, had been tested in follow-up studies to establish if, in fact, the fish population in the streams had increased or decreased as a result of the new flows" (Hatfield and Howard 1983). And this statement was made after recent interviews with 27 developers or users of incremental-type methodologies (Appendix 1, Table 2). The relationship between fish abundance and weighted usable area has been abandoned as an issue by IFIM and is dealt with solely by implicit assumption even though this is the ultimate objective of the development and use of this method. Clearly then, the greatest weakness of IFIM is its current lack of testing, validation, and follow-up assessments despite its hundreds of applications.

In addition to the poor relationship between fish biomass and weighted usable area, the similar but separate problem is that in most applications, WUA is not calibrated with fish abundance or biomass. The consequences of this is that PHABSIM analyses (i.e. changes in WUA) cannot be used to make streamflow recommendations to protect fish. Making a streamflow recommendation from a calculation of WUA is equivalent to having only the x coordinates of several coordinate pairs of a regression equation and then being asked, given a new x what is the new y? If the slope is not 1 and the y-intercept is not 0 (probably the case of both) then it is impossible to draw credible conclusions about changes in y resulting from changes in x. Often in

streamflow alterations the statistic of interest is not the change in y at the new x, but the magnitude of y at the new x. This is a bit more difficult to predict than the change in y, and the prediction is even less credible. Deriving a flow recommendation from only a calculation of WUA without knowing the slope, the y intercept, or the y's of the coordinate pairs is nonsense and has no solvable answer. However, this is exactly what many applications of PHABSIM attempt to do, and is equivalent to an intuitive assumption of regression.

This error in assumption results in misuse of the analysis. "The WUA...cannot be used to estimate populations which might be sustained without field data to correlate the WUA with fish biomass or numbers" (Hatfield and Howard 1983). This misuse arises from the general failure of PHABSIM users to realize that the method deals only with the relationship between streamflow and the hydraulic variables (relationship 2; Fig. 1). Most users uncritically assume that the method also deals with the relationship between fish habitat and population size (relationship 3; Fig. 1). It does not. "The Instream Flow Incremental Methodology does not produce an instream flow recommendation" (Hatfield and Howard 1983 (underling mine)), yet the objective of every application of the method is to produce such a recommendation. "The IFG methodology...cannot be used to estimate actual usable habitat for conditions other than those actually observed because the WUA values obtained by the IFG approach are not properly normalized to total number of fish" (Hatfield and Howard 1983). The consequence of weighted usable area not being calibrated to fish abundance is that streamflow recommendations required to maintain a desired fish abundance or biomass cannot be made from PHABSIM analyses.

THE DEFINITION OF HABITAT

Some researchers feel the cause for poor relationships between WUA (= habitat) and fish populations is that fish "habitat" as defined and calculated by PHABSIM is imperfectly defined. "WUA values...do not provide an...index of...habitat...available.... As a result...IFG flow recommendations ...are unreliable" (Nelson 1980a). If fish "habitat" as it is calculated by PHABSIM is faulty in considering all variables which determine the size of fish populations, then relationships between fish "habitat" and population abundance or biomass would be expected to be weak or obscure (for example some of the analyses in Table 2). "The absence of well-defined relationships between the calculated WUA and the standing crops of adult brown and rainbow trout...is not unexpected (because) the WUA-discharge relationships derived for the study reaches do not depict the actual relationships between adult trout habitat and (stream) flow" (Nelson 1980a). Other variables such as water temperature (Ferguson 1958), dissolved oxygen (Whitmore, Warren and Doudoroff 1960), pH (Haines 1981), suspended solids (Cordone and Kelly 1961), nitrate (Binns and Eiserman 1979), phosphate (Jones and Hoyer 1982), and calcium (McFadden and Cooper 1962) affect the distribution, abundance, and biomass of fish. This being so, they are properly considered as and included in calculations of fish habitat.

THE EXISTENCE OF "OTHER VARIABLES"

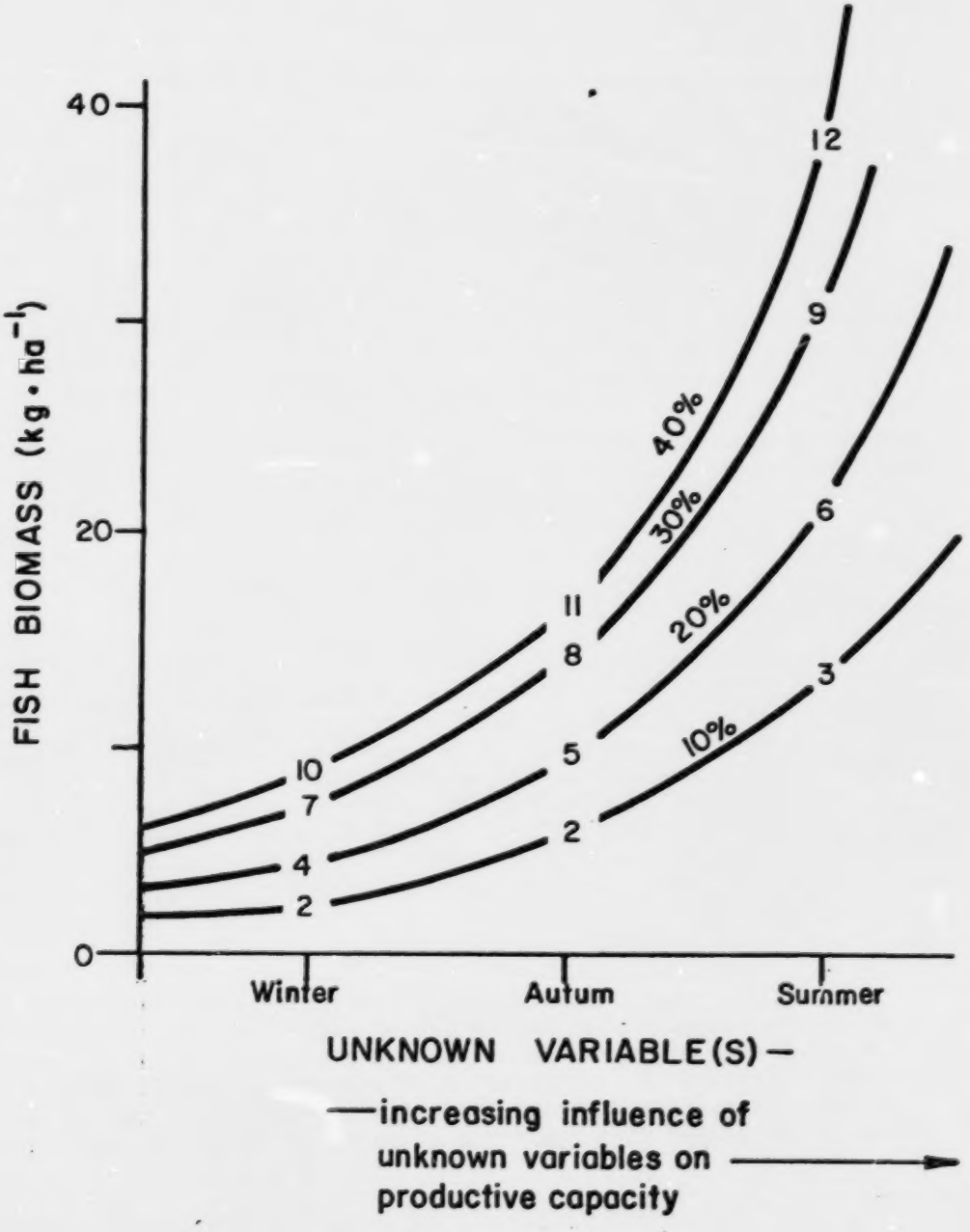
It is possible to demonstrate the existence of factors other than weighted usable area (water depth, water velocity, substrate size) which influence the biomass of fish populations. Figure 13, drawn from published data calculated using the Instream Flow Incremental Methodology (Orth and Maughan 1982), shows the effect on population biomass of increasing both the amount of weighted usable area and increasing other important, but unknown variables. It is likely that these are water temperature, dissolved oxygen, turbidity, nitrate, phosphate, and calcium, but it is possible to discuss the existence of these variables without being certain about their identity. Let us call them "unknown variables".

There are three important conclusions that can be drawn from Figure 13 about these "unknown variables". The first and most important is that they exist - that is, variables other than depth, velocity, and substrate (weighted usable area) are playing important roles in determining the biomass of the population. The evidence for this is that increases in the amount of weighted usable area shift the biomass curve upwards (differences in biomass between lines) but equally large increases in biomass occur with changes in other "unknown variables" (difference in biomass along lines). (The seasons appear out of sequence the way I have drawn them in Figure 13, however, this is the correct order for ranking their ascending influence.) For example, when 10% of the stream area is usable habitat, in summer it supports 10 kilograms of fish per hectare ($kg \cdot ha^{-1}$), in autumn it supports 5 $kg \cdot ha^{-1}$, and in winter it supports 2 $kg \cdot ha^{-1}$. This is a five-fold increase in biomass with no change in the percent of weighted usable area. Likewise, when 10% of the stream area in summer is usable habitat, it can support a higher biomass of fish than when 40% of the stream area is usable in winter. Apparently, an unknown but normal increase in "unknown variables" occurs seasonally from winter to summer, and has the same effect on population biomass as increasing the amount of weighted usable area four times. Clearly something in addition to the amount of usable habitat (weighted usable area) is determining the biomass of the population.

A second important conclusion about the "unknown variables", regardless of what the variables are or what their levels are in different seasons, is that their influence has a greater positive effect in summer than in winter (i.e. the influence of the variables on population biomass does not change linearly over the year; the slopes of the regression lines change depending upon the season. Note the distinction I am making between the size of the biomass and the rate at which the biomass responds to increases in "unknown variables": at higher levels of influences of "unknown variables" not only is the biomass greater (compare biomass at point 12 with biomass at point 10), but the slope of the regression line is steeper (compare slope at point 12 with slope at point 10).

A third and final important conclusion about the "unknown variables" is that they interact with the amount of usable habitat in their influence on the population's biomass. For example the slope at point 10 is steeper than at point 1 because of an increase in the amount of usable habitat (between line comparisons of 40% versus 10% weighted usable area) while the slope at point 12 is steeper than the slope at point 10 because of an increase in the

Fig. 13. Biomass of central stoneroller (Campostoma anomalum) in Glover Creek under a constant amount (10%-40%) of weighted usable area over seasons (adapted from Orth and Maughan 1982; Fig. 8). Numbers on the isopleths are points on the curves discussed in the text.



influence of "unknown variables" (along line comparisons of summer versus winter conditions). However, the slope at point 12 is steeper than the slope at either point 1 or point 10 because both more habitat is available, and the influence of "unknown variables" is greater. The significance of this interaction is important because it means greater amounts of usable habitat (weighted usable area) allow a greater response of population biomass to increases in "unknown variables" (i.e. the amount of usable habitat constrains the response of population biomass to increase in "unknown variables").

Experimental evidence for the existence of "unknown variables" is provided by a series of conrtolled studies which related weighted usable area to the biomass of juvenile rainbow trout. White et al. (1981, p. 240) found that "...factors (= my unknown variables?) other than depth and velocity were operating to bring about the (reduction in trout biomass), since calculated WUA actually increased at this (reduced) flow. Other discharge reductions tested resulted in fish...not decreasing as sharply as predicted". Binns and Eiserman (1979) developed models relating trout standing crop to eight habitat variables which were significantly correlated with trout biomass. In addition to the physical habitat variables used in PHABSIM, several other variables related to the productive capacity of the environment (e.g. maximum summer water temperature, nitrate nitrogen, fish food abundance, and fish food diversity). These may be the "unknown variables" affecting fish biomass in Orth and Maughan's (1982) and White et al.'s (1981) studies. Let us assume that the "unknown variables" influence the "productive capability" of a stream environment.

WHICH IS MORE IMPORTANT: WUA or "PRODUCTIVE CAPABILITY"

Having identified the existence of variables other than weighted usable area which affect a population's biomass, it is possible to compare the importance of changes in these other variables relative to changes in the amount of weighted usable area for determining fish biomass. For the data available (i.e. Fig. 13) changes in "productive capability" give a greater response in fish biomass than changes in the amount of weighted usable area. For example, when low levels of stream area are usable (10%), a normal increase in "productive capability" gives a greater response in population biomass than does a four-fold increase in usable space (compare the difference in biomass between point 3 and point 1 - the increase due to "productive capability" with the difference in biomass between points 10 and 1 - the increase due to the amount of WUA). Likewise at high "productive capability", a reduction in "productive capability" gives a greater response in population biomass than does a 75% reduction in usable area (compare the difference in biomass between point 12 and point 10 - the decrease due to reduction in "productive capability" with the difference in biomass between points 12 and 3 - the decrease due to reduction in WUA). In both cases, the change in population biomass due to changes in "productive capability" exceeded the change due to changes in weighted usable area. It is important to qualify these results by noting that they occurred with limited amount of the stream being usable habitat (a maximum of 40%), however, based on comparisons, within this range, "productive capability" is <u>more important</u> than weighted usable area in determining the biomass of a population.

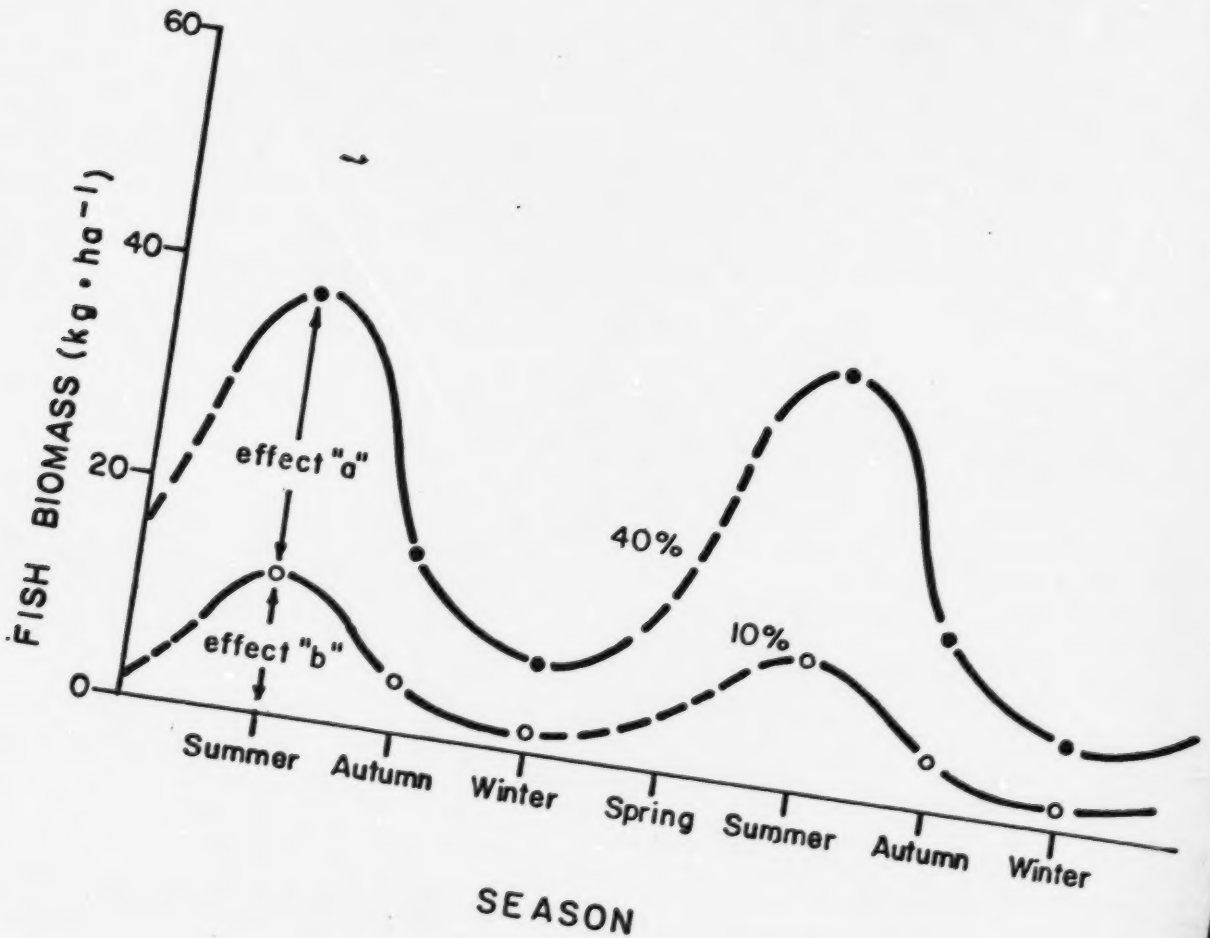
THE MECHANISM OF INFLUENCE

The way reductions in weighted usable area affect fish populations appears to be that the amount of usable habitat constrains the response of a population's biomass to changes in "productive capability". At low levels of "productive capability" increases in weighted usable area have relatively little effect on population biomass, while at high levels of "productive capability" increases in weighted usable area result in a greater response in population biomass (Fig. 13). In other words, a greater amount of weighted usable area allows the population to respond to increases in "productive capability" faster and to a greater extent (i.e. the slope of the regression lines progressively increase as more of the stream area becomes usable). It appears from these responses (Fig. 13) that the amount of weighted usable area is a constraint which limits the size of the population response to an increase in "productive capability". Reductions in streamflow reduce the biomass of the population. Let us call the response "effect A" (Fig. 14) and define it as the "level of constraint" caused by the amount of weighted usable area.

The way changes in "productive capability" affect a population's biomass is different from the way changes in weighted usable area does. By redrawing data from Orth and Maughan (1982) it can be shown that when the percent of usable habitat (weighted usable area) is constant, the biomass of the population has an annual cycle which peaks in summer and is lowest in winter (Fig. 14). The pattern of the biomass cycle and the size of the biomass at any time through the year depends upon whether or not the mechanisms which increase biomass exceed the mechanisms which decrease biomass. Recruitment (hatching, immigration) and growth increase a populations's biomass while emigration, and death and predation decrease a population's biomass. An impact which affects any or all of recruitment, growth, emigration, or death will influence both the pattern of the biomass cycle and the size of the biomass. It is important to recognize that hatching, growth, and death are largely metabolic functions, while immigration and emigration are behavioral functions. Because changes in water depth and water velocity affect metabolism as well as behavior, changes in streamflow can alter the shape and amplitude of the biomass cycle. (Note here, however, that in some populations predation, both natural and man-caused, can play a major role in annual biomass cycles, and that changes in streamflow, especially through changes in water depth, can alter the degree and duration that populations are exposed to exploitation.)

Changes in the productive capability of an environment are partly responsible for causing the annual cycle in a population's biomass. Habitat variables which determine an environment's productive capability include water temperature, dissolved oxygen, pH, nitrate, and phosphate. Because changes in streamflow affect these variables, and they in turn affect metabolism, the biomass cycle of fish may be depressed or eliminated by changes in streamflow. Let us call such a potential response to changes in streamflow "effect B" (Fig. 14), and define it as the reduction in biomass caused by changes in streamflow affecting metabolism.

Fig. 14. Annual biomass cycle of central stoneroller (Campostoma anomalum) in Glover Creek when 10% and 40% of the habitat is usable (adapted from Orth and Maughan 1982; Fig. 8). Solid lines are data from Orth and Maughan 1982; broken lines are assumed interpolations I have made.



THE RELEVANCE OF WUA

PHABSIM may be a method of calculating the "level of constraint" caused by reductions in weighted usable area but it does not calculate how changes in streamflow affect the productive capability of the environment. This is important to the success of the Instream Flow Incremental Methodology because if changes in streamflow affect either the rate or extent that the productive capability changes normally, then the biomass response of the population may be determined to a much greater extent indirectly by the effects of streamflow on metabolism than by the direct effect of streamflow on the amount of weighted usable area. If the mechanism of the effect of reductions in weighted usable area on fish biomass is "constraint" only (and this is apparently the way most users of the Instream Flow Incremental Methodology are interpreting WUA), then in PHABSIM all other factors affecting fish production are assumed not to change. (That is, the normal cycle of biomass continues but at lower levels of biomass. The factors causing biomass to cycle are assumed to be unchanged.) This is unlikely because changes in water depth and water velocity affect metabolism as well as behavior. The real effect of reduced streamflow is likely the sum of the reduction in weighted usable area plus reductions in metabolism. The current failure of fisheries science to effectively deal with regulated streamflows is caused by our lack of knowledge of how changes in streamflow are related to the productive capability of the environment. Nevertheless, metabolic effects have to be incorporated into any model which is to successfully predict consequences of streamflow alteration to fish populations.

Because weighted usable area does not include any measure of the productive capability of the environment, weighted usable area should not be thought of synonymously as habitat. This one assumption (i.e. that WUA = habitat) is the kingpin of the rationale of IFIM and it is false (Fig. 15). If WUA is irrelevant for recommending streamflows to protect fish (because of its poor relationship with fish abundance or biomass), then PHABSIM is irrelevant because WUA is the output of PHABSIM. If PHABSIM is irrelevant, then IFIM is irrelevant because PHABSIM is IFIM[1].

It is clear that for some weighted usable area has no meaning as fish habitat. "Values of WUA obtained by the IFG methods <u>might</u> provide <u>some kind</u> of index (of habitat) under <u>some</u> condition" (Hatfield and Howard 1983 (underlining mine)). These authors seem to be saying that WUA is a meaningless calculation - meaningless not in the sense that the calculation is futile, but that once it is made, its significance is uncertain. Using this connotation, I agree that WUA is a meaningless calculation. However, I would go further than that and suggest that users who derive flow recommendations following the procedure of intuitive regression assumption, then WUA is not only meaningless, but can be misleading. That is, agencies charged with

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[1]This is debatable, but only in theory. IFIM contains two other programs of equal importance to PHABSIM: SSAMIV - the water quality model, and IWTM - the water temperature model, but because these two models have never been included in any of the published IFIM studies (Table 2) and since they are apparently rarely used in any of the many unpublished applications of IFIM, as a consequence PHABSIM is the only part of IFIM in widespread use. If effect, then, PHABSIM is IFIM.

INSTREAM FLOW RELATIONSHIPS

PROBLEM: water removed for irrigation
industrial/domestic supply
hydroelectric generation

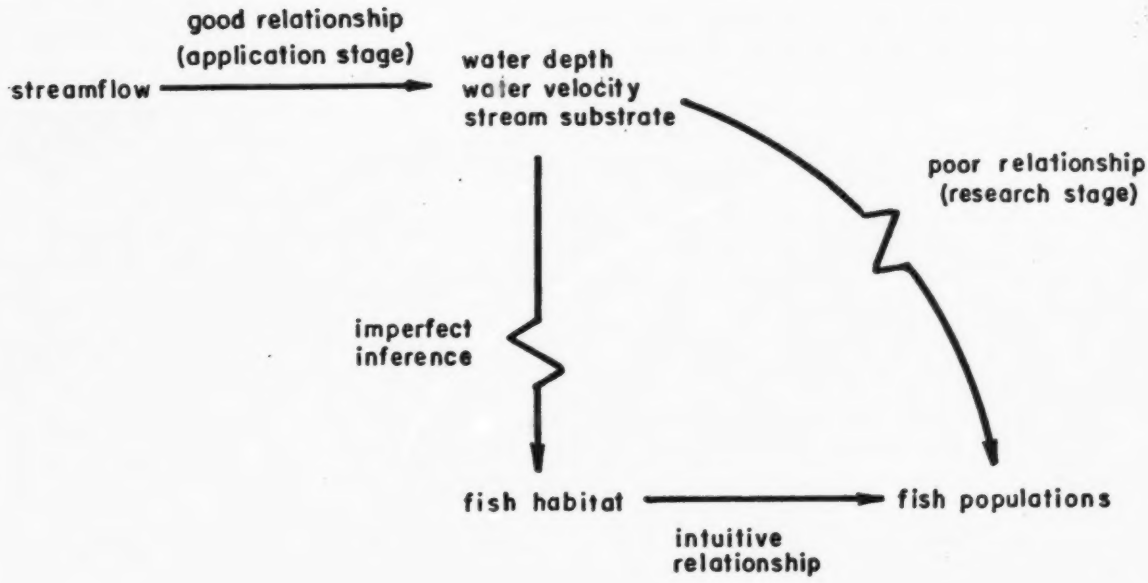

Fig. 15. Current status of the progress made towards determining the relationship between streamflow and fish populations. (Compare with Fig. 1, the starting model for these relationships).

managing fish populations and their habitat, and who use inferences based on calculations of WUA to reach streamflow recommendations for protecting those populations, could made decisions which may actually take them away from their objective of protecting fish.

CONCLUSIONS

1) In the calculation of WUA there is a possibility of large error due to the way water depth and water velocity are calculated and the way the habitat suitability curves are interpreted.

2) Validation studies done to date show an inconsistent relationship between weighted usable area and fish abundance and biomass, therefore a positive linear relationship between them should not be assumed.

3) Because there are only few follow-up studies on the effect of streamflows recommended by IFIM analyses its effectiveness for protecting fish populations is as yet unproven.

4) Other variables in addition to water depth, water velocity and substrate size are important in determining fish abundance and biomass.

5) Metabolic effects due to altered streamflows (productive capability of the environment) may be more important than behavioral effects due to changes in weighted usable area.

6) WUA is not equivalent to habitat and alone could be misleading as to the effect of a streamflow alteration.

7) PHABSIM analysis alone is insufficient for complete impact assessment of streamflow alteration projects and should no longer be accepted as the sole criterion for streamflow recommendations.

8) Caution should be used when applying PHABSIM until it receives further testing, validation, and modification.

9) The metabolic effect of reduced streamflows should be incorporated into methods used to assess streamflow impacts and fish production should be used as the criterion of impact instead of fish abundance or biomass.

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APPENDIX 1

Personal Communications Cited in the Text

Table 1. List of contacts for personal communications cited in the text.

Name	Agency	Mailing Address	Telephone Number
1. Alan Elser	Montana Department of Fish, Wildlife and Parks Fisheries Services Division.	1420 East 6th Ave. Helena, Montana 59620	406-444-3183
2. Dave Morantz	Canada Department of Fisheries and Oceans.	P.O. Box 550 Halifax, Nova Scotia B3J 2S7	902-426-2480
3. Donald Orth	Virginia Polytechnic Institute and State University, Department of Fisheries and Wildlife Sciences.	156 Cheatham Hall Blacksburg, Virginia 24061	703-961-5919

Table 2. List of contacts for personal communications cited by Hatfield and Howard (1983).

Name	Agency	Address	Telephone Number	Telephone Interview	In Person Interview
1. Tom Annear	Wyoming Game and Fish Department.	5400 Bishop Blvd. Cheyenne, Wyoming 82002	307-777-7686		X
2. Allan Ansell	Idaho Power Company.	P.O. Box 70 Boise, Idaho 83707	208-353-2720		X
3. Hal Beecher	Washington Department of Game.	Olympia, Washington	206-753-2737		X
4. Ken Bovee	Cooperative Instream Flow Services Group.	Creekside Building 2625 Redwing Rd. Fort Collins, Colorado 80526	303-226-9331		X
5. C.B. Burton	Utah Power and Light Co.	1407 West North Temple P.O. Box 899 Salt Lake City, Utah	801-535-2000		X
6. Tim Cochnauer	Idaho Department of Fish and Game.	2320 Government Way Coeur d'Alene, Idaho 83814	208-664-9236		X
7. Al Conder	Wyoming Game and Fish Department.	5400 Bishop Blvd. Cheyenne, Wyoming 82002	307-777-7686	X	X
8. W. Eifert	Water Resources Research Institute.	Room 227, Wyoming Hall P.O. Box 3067 University of Wyoming Laramie, Wyoming 82071	307-766-2143		X

Table 2. (cont'd.)

	Name	Agency	Address	Telephone Number	Telephone Interview	In Person Interview
9.	J. Esch	National Marine Fisheries Service.	Portland, Oregon	503-230-5430		X
10.	S.K. Evans	Utah Power and Light Co.	1407 West North Temple P.O. Box 899 Salt Lake City, Utah	801-535-2000		X
11.	Dick Fisher	Cooperative Instream Flow Services Group.	Creekside Building 2625 Redwing Rd. Fort Collins, Colorado 80526	303-226-9331		X
12.	Bill Geer	Utah Division of Wildlife Resources.	1596 West North Temple Salt Lake City, Utah 84116	801-533-9333	X	X
13.	R. Gerke	Washington Department of Fisheries.	115 General Administration Building Olympia, Washington 98504	206-753-6600		X
14.	R. Hamilton	Department of Fisheries and Oceans.	1090 West Pender Street Vancouver, B. C.		X	
15.	Bill Horton	Idaho Department of Fish and Game.	1540 Warner Avenue Lewiston, Idaho 83501	208-743-6502	X	X
16.	I.W. Jones	Oregon Department of Fish and Wildlife.	506 S.W. Mill Street P.O. Box 3503 Portland, Oregon 97208	503-229-5683	X	X

Table 2. (cont'd.)

	Name	Agency	Address	Telephone Number	Telephone Interview	In Person Interview
17.	Ed Kochman	Colorado Division of Wildlife.	6060 Broadway Denver, Colorado	303-297-1192	X	
18.	Lee Lamb	Cooperative Instream Flow Services Group.	Creekside Building 2625 Redwing Rd. Fort Collins, Colorado 80526	303-226-9331		X
19.	Robert M. Lee	Portland General Electric.	121 S.W. Salmon Street Portland, Oregon	503-226-8370		X
20.	Bob Milhous	Cooperative Instream Flow Services Group.	Creekside Building 2625 Redwing Rd. Fort Collins, Colorado 80526	303-226-9331		X
21.	LeRoy E. Newland	Utah Power and Light Co.	1407 West North Temple P.O. Box 899 Salt Lake City, Utah	801-535-2000		X
22.	L.S. Pearson	Oregon Department of Fish and Wildlife.	506 S.W. Mill Street P.O. Box 3503 Portland, Oregon 97208	503-229-5683		X
23.	Larry Peterman	Montana Department of Fish, Wildlife & Parks Ecological Services Division.	1420 E. 6th Avenue Helena, Montana	406-449-3888	X	
24.	Ken Slattery	Washington Department of Ecology.	Olympia, Washington	206-459-6114		X

Table 2. (cont'd.)

	Name	Agency	Address	Telephone Number	Telephone Interview	In Person Interview
25.	Clair Stalnaker	Cooperative Instream Flow Services Group.	Creekside Building 2625 Redwing Rd. Fort Collins, Colorado 80526	303-226-9331	X	X
26.	Gerald Swank	U.S. Forest Service.	319 S.W. Pine Street Portland, Oregon	503-221-3032	X	X
27.	Thomas A. Wesche	Water Resources Research Institute.	Room 227, Wyoming Hall P.O. Box 3067 University of Wyoming Laramie, Wyoming 82071	307-766-2143	X	

APPENDIX 2

Derivation of Data Used by Analysis

In the Text

Table 1. Error of the velocity measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the mean cell width is 1/30 the mean river width.

Measurement	Distance between measurements (m)	Velocity ($m \cdot s^{-1}$)	Difference in velocity between measurements ($m \cdot s^{-1}$)	½ difference in velocity between measurements as %[a]
1	0.30	-0.09		
			0.36	200.0
2	0.30	0.27		66.6
			0.66	122.2
3	0.30	0.93		35.5
			0.06	3.2
4	0.30	0.86		3.5
			0.01	0.5
5	0.30	0.85		0.6
			0.04	2.4
6	0.30	0.89		2.3
			0.10	5.6
7	0.30	0.99		5.1
			0.12	6.1
8	0.30	0.87		6.9
			0.0	0.0
9	0.30	0.87		0.0
			0.07	4.0
10	0.30	0.94		3.7
			0.09	4.8
11	0.30	1.05		4.3
			0.06	2.9
12	0.30	0.99		3.0
			0.15	7.8
13	0.30	0.84		8.9
			0.12	7.1
14	0.30	0.72		8.3
			0.03	2.1
15	0.30	0.69		2.2
			0.09	6.5
16	0.30	0.78		5.8
			0.27	17.3
17	0.30	0.51		26.5
			0.03	2.9
18	0.30	0.49		3.1
			0.04	4.1
19	0.30	0.53		3.8
			0.08	7.6
20	0.30	0.45		8.9
			0.03	3.3
21	0.30	0.48		3.1
			0.07	7.3
22	0.30	0.41		8.5
			0.04	4.9
23	0.30	0.37		5.4
			0.02	2.4
				2.9

Table 1 (cont'd)

Measurement	Distance between measurements (m)	Velocity ($m \cdot s^{-1}$)	Difference in velocity between measurements ($m \cdot s^{-1}$)	½ difference in velocity between measurements as %[a]
24	0.30	0.35		1.4
			0.01	1.5
25	0.30	0.34		23.5
			0.16	44.4
26	0.30	0.18		27.8
			0.10	17.9
27	0.30	0.28		14.3
			0.08	20.0
28	0.30	0.20		50.0
			0.20	n = 55
29	0.30	0.0		$\bar{x}$ = 15.4
			0.0	SD = 32.3
30	0.30	0.0		

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 2. Error in the velocity measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the mean cell width is 1/15 the mean river width.

Measurement	Distance between measurements (m)	Velocity ($m \cdot s^{-1}$)	Difference in velocity between measurements ($m \cdot s^{-1}$)	½ difference in velocity between measurements as %[a]
1	0.60	0.27		109.2
			0.59	34.3
2	0.60	0.86		1.7
			0.03	1.7
3	0.60	0.89		1.1
			0.02	1.2
4	0.60	0.87		4.0
			0.07	3.7
5	0.60	0.94		2.7
			0.05	2.5
6	0.60	0.99		13.6
			0.27	18.8
7	0.60	0.72		4.2
			0.06	3.9
8	0.60	0.78		18.6
			0.29	29.6
9	0.60	0.49		4.1
			0.04	4.4
10	0.60	0.45		4.4
			0.04	4.9
11	0.60	0.41		7.3
			0.06	8.6
12	0.60	0.35		24.3
			0.17	47.2
13	0.60	0.18		5.6
			0.02	5.0
14	0.60	0.20		50.0
			0.20	n = 27
15	0.60	0.0		$\bar{x}$ = 15.4
				SD = 23.3

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 3. Error of the velocity measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the mean cell width is 1/8 the mean river width.

Measurement	Distance between measurements (m)	Velocity ($m \cdot s^{-1}$)	Difference in velocity between measurements ($m \cdot s^{-1}$)	½ difference in velocity between measurements as %[a]
1	1.2	0.27		114.8
			0.62	34.8
2	1.2	0.89		2.8
			0.05	2.7
3	1.2	0.94		11.7
			0.22	15.3
4	1.2	0.72		16.0
			0.23	23.5
5	1.2	0.49		8.2
			0.08	9.8
6	1.2	0.41		28.1
			0.23	63.9
7	1.2	0.18		50.0
			0.18	n = 13
8	1.2	0.0		$\bar{x}$ = 29.4
				SD = 31.5

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 4. Error of the depth measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the ratio of mean cell width to mean river width is 1/30.

Measurement	Distance between measurements (m)	Depth (m)	Difference in depth between measurements (m)	⅓ difference in depth between measurements as %[a]
1	0.30	0.04		337.5
			0.27	43.6
2	0.30	0.31		9.7
			0.06	8.1
3	0.30	0.37		5.4
			0.04	4.9
4	0.30	0.41		1.2
			0.01	1.2
5	0.30	0.42		1.2
			0.01	1.2
6	0.30	0.43		0.0
			0.00	0.0
7	0.30	0.43		0.0
			0.00	0.0
8	0.30	0.43		1.2
			0.01	1.2
9	0.30	0.42		2.4
			0.02	2.5
10	0.30	0.40		3.8
			0.03	4.1
11	0.30	0.37		4.1
			0.03	4.4
12	0.30	0.34		5.9
			0.04	6.7
13	0.30	0.30		5.0
			0.03	5.6
14	0.30	0.27		7.4
			0.04	8.7
15	0.30	0.23		6.5
			0.03	7.5
16	0.30	0.20		0.0
			0.00	0.0
17	0.30	0.20		0.0
			0.00	0.0
18	0.30	0.20		5.0
			0.02	5.6
19	0.30	0.18		5.6
			0.02	6.3
20	0.30	0.16		9.4
			0.03	11.5
21	0.30	0.13		11.5
			0.03	15.0
22	0.30	0.10		10.0
			0.02	8.3
23	0.30	0.12		4.2
			0.01	3.9

Table 4 (cont'd)

Measurement	Distance between measurements (m)	Depth (m)	Difference in depth between measurements (m)	⅓ difference in depth between measurements as %[a]
24	0.30	0.13		0.0
			0.00	0.0
25	0.30	0.13		3.9
			0.01	4.2
26	0.30	0.12		4.2
			0.01	4.6
27	0.30	0.11		4.6
			0.01	5.0
28	0.30	0.10		20.0
			0.04	33.3
29	0.30	0.06		33.3
			0.04	100.0
30	0.30	0.02		n = 58
				$\bar{x}$ = 13.8
				SD = 45.7

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 5. Error of the depth measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when ratio of mean cell width to mean river width is 1/15.

Measurement	Distance between measurements (m)	Depth (m)	Difference in depth between measurements (m)	½ difference in depth between measurements as %[a]
1	0.60	0.31		16.1
			0.10	12.2
2	0.60	0.41		2.4
			0.02	2.3
3	0.60	0.43		0.0
			0.00	0.0
4	0.60	0.43		3.5
			0.03	3.8
5	0.60	0.40		7.5
			0.06	8.8
6	0.60	0.34		10.3
			0.07	13.0
7	0.60	0.27		13.0
			0.07	17.5
8	0.60	0.20		0.0
			0.00	0.0
9	0.60	0.20		10.0
			0.04	12.5
10	0.60	0.16		18.8
			0.06	30.0
11	0.60	0.10		15.0
			0.03	11.5
12	0.60	0.13		3.9
			0.01	4.2
13	0.60	0.12		8.3
			0.02	10.0
14	0.60	0.10		40.0
			0.08	200.0
15	0.60	0.02		n = 28
				$\bar{x}$ = 17.0
				SD = 37.0

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 6. Error of the depth measurements along transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the ratio of mean cell width to mean river width is 1/8.

Measurement	Distance between measurements (m)	Depth (m)	Difference in depth between measurements (m)	½ difference in depth between measurements as %[a]
1	1.2	0.31		19.4
			0.12	14.0
2	1.2	0.43		3.5
			0.03	3.8
3	1.2	0.40		16.3
			0.13	24.1
4	1.2	0.27		13.0
			0.07	17.5
5	1.2	0.20		25.0
			0.10	50.0
6	1.2	0.10		10.0
			0.02	8.3
7	1.2	0.12		41.7
			0.10	250.0
8	1.2	0.02		n = 14
				$\bar{x}$ = 35.5
				SD = 63.1

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 7. Error in the velocity measurements downstream from transect 1 of the Pembroke River (Shirvell & Horantz 1983) when the distance between transects is 0.8 river widths.

Measurement	Distance between measurements(m)	Velocity at transect 1 ($m \cdot s^{-1}$)	Velocity at transect 2 ($m \cdot s^{-1}$)	Difference in velocity between transects 1 and 2 ($m \cdot s^{-1}$)	½ difference in velocity between transects as %[a]
1	7.5	-0.09	0.14	0.25	138.9
2	7.5	0.27	0.14	0.13	89.3 24.1
3	7.5	0.93	0.25	0.68	46.4 36.6
4	7.5	0.85	1.34	0.49	136.0 28.8
5	7.5	0.89	1.47	0.58	18.3 32.6
6	7.5	0.99	0.98	0.01	19.7 0.5
7	7.5	0.87	1.19	0.32	0.4 18.4
8	7.5	0.94	1.42	0.48	13.5 25.5
9	7.5	1.05	1.34	0.27	16.9 12.9
10	7.5	0.84	1.06	0.24	10.1 14.3
11	7.5	0.72	1.27	0.55	11.3 34.0
12	7.5	0.69	0.93	0.24	19.3 17.4
13	7.5	0.51	1.01	0.50	12.9 49.0
14	7.5	0.49	0.63	0.14	24.8 14.3
15	7.5	0.53	0.80	0.27	11.1 25.5
16	7.5	0.48	0.72	0.24	16.9 25.0
17	7.5	0.41	0.43	0.02	16.7 2.4
18	7.5	0.37	0.33	0.04	2.3 5.4
19	7.5	0.34	0.57	0.13	6.1 19.1
20	7.5	0.18	0.24	0.06	11.4 16.7
21	7.5	0.28	0.08	0.20	12.5 35.7
22	7.5	0.00	0.00	0.00	125.0 0.0 0.0
					n = 44 $\bar{x}$ = 27.2 SD = 33.1

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 8. Error in the velocity measurements downstream from transect 1 of the Pembroke River (Shirvell & Horants 1983) when the distance between transects is 2.7 river widths.

Measurement	Distance between measurements(m)	Velocity at transect 1 ($m \cdot s^{-1}$)	Velocity at transect 3 ($m \cdot s^{-1}$)	Difference in velocity between transects 1 and 3 ($m \cdot s^{-1}$)	¼ difference in velocity between transects as %[a]
1	24.0	-0.09	0.27	0.36	200.0 66.7
2	24.0	0.27	0.72	0.45	83.3 31.3
3	24.0	0.93	0.45	0.48	25.8 53.3
4	24.0	0.85	0.48	0.37	21.8 38.5
5	24.0	0.89	0.62	0.27	15.2 21.8
6	24.0	0.99	0.35	0.64	32.3 91.4
7	24.0	0.87	0.40	0.47	27.0 58.8
8	24.0	0.94	0.54	0.50	26.6 46.3
9	24.0	1.05	0.37	0.68	32.4 91.9
10	24.0	0.84	0.42	0.42	25.0 50.0
11	24.0	0.72	0.50	0.22	15.3 22.0
12	24.0	0.69	0.36	0.33	23.9 45.8
13	24.0	0.51	0.21	0.30	29.4 71.4
14	24.0	0.49	0.21	0.28	28.6 66.7
15	24.0	0.53	0.26	0.27	25.6 51.9
16	24.0	0.48	0.32	0.16	16.7 25.0
17	24.0	0.41	0.19	0.22	26.8 57.9
18	24.0	0.37	0.15	0.22	29.7 73.3
19	24.0	0.34	0.11	0.23	33.8 104.6
20	24.0	0.18	0.07	0.11	30.6 78.6
21	24.0	0.28	0.14	0.14	25.0 50.0
22	24.0	0.00	0.20	0.20	n = 42 $\bar{x}$ = 47.0 SD = 33.7

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.

Table 9. Error of the depth measurements downstream from transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the distance between transects is 0.8 river widths.

Measurement	Distance between measurements (m)	Depth (m) at transect 1	Depth (m) at transect 2	Difference in depth between transects 1 and 2 (m)	½ difference in depth between transects as %[a]
1	7.5	0.04	0.71	0.67	837.5 47.2
2	7.5	0.31	0.72	0.41	66.1 28.5
3	7.5	0.37	0.71	0.34	46.0 23.9
4	7.5	0.42	0.70	0.28	33.3 20.0
5	7.5	0.43	0.68	0.25	29.1 18.4
6	7.5	0.43	0.66	0.23	26.7 17.4
7	7.5	0.42	0.62	0.20	23.8 16.1
8	7.5	0.40	0.57	0.17	21.3 14.9
9	7.5	0.37	0.52	0.15	20.3 14.4
10	7.5	0.30	0.45	0.15	25.0 16.7
11	7.5	0.27	0.42	0.15	27.8 17.9
12	7.5	0.23	0.39	0.16	34.8 20.5
13	7.5	0.20	0.36	0.16	40.0 22.2
14	7.5	0.20	0.32	0.12	30.0 18.8
15	7.5	0.18	0.26	0.08	22.2 15.4
16	7.5	0.13	0.21	0.08	30.8 19.1
17	7.5	0.10	0.19	0.09	45.0 23.7
18	7.5	0.12	0.17	0.05	20.8 14.7
19	7.5	0.13	0.16	0.03	11.5 9.4
20	7.5	0.12	0.15	0.03	12.5 10.0
21	7.5	0.11	0.10	0.01	4.6 5.0
22	7.5	0.06	0.05	0.01	8.3 10.0
					n = 44 $\bar{x}$ = 41.4 SD =123.4

[a]Each number of the pair for each station represents the calculated error at either side of the cell.

Table 10. Error of the depth measurements downstream from transect 1 of the Pembroke River (Shirvell & Morantz 1983) when the distance between transects is 2.7 river widths.

Measurement	Distance between measurements (m)	Depth at transect 1 (m)	Depth at transect 3 (m)	Difference in depth between transects 1 and 3 (m)	½ difference in depth between transects as %[a]
1	24.0	0.04	0.69	0.65	812.5 47.1
2	24.0	0.31	0.79	0.48	77.4 30.4
3	24.0	0.37	0.87	0.50	67.6 28.7
4	24.0	0.42	0.95	0.53	63.1 27.9
5	24.0	0.43	0.95	0.52	60.5 27.4
6	24.0	0.43	0.94	0.51	59.3 26.7
7	24.0	0.42	1.03	0.61	72.6 29.6
8	24.0	0.40	1.14	0.74	92.5 32.5
9	24.0	0.37	1.16	0.79	106.8 34.1
10	24.0	0.30	1.07	0.77	111.7 31.3
11	24.0	0.27	0.98	0.71	131.5 31.1
12	24.0	0.23	0.91	0.68	147.8 37.4
13	24.0	0.20	0.84	0.64	160.0 38.1
14	24.0	0.20	0.80	0.60	150.0 37.5
15	24.0	0.18	0.73	0.55	152.8 37.7
16	24.0	0.13	0.72	0.59	226.9 41.0
17	24.0	0.10	0.63	0.53	265.0 42.1
18	24.0	0.12	0.54	0.42	175.0 38.9
19	24.0	0.13	0.45	0.32	123.1 35.6
20	24.0	0.12	0.37	0.25	104.2 33.8
21	24.0	0.11	0.29	0.18	81.8 31.0
22	24.0	0.06	0.12	0.06	50.0 25.0
					n = 44 $\bar{x}$ = 91.8 SD =125.1

[a] Each number of the pair for each measurement represents the calculated error at either side of the cell.